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PROCEEDINGS OF A WORKSHOP ON ASSESSING ABUNDANCE AND TRENDS FOR IN-WATER SEA TURTLE POPULATIONS

Held at the Archie Carr Center for Sea Turtle Research
University of Florida, Gainesville, Florida
24 - 26 March 2000

Edited by

Karen A. Bjorndal
Alan B. Bolten

October 2000

U. S. Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149

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Of course, the success of the workshop rests largely with the participants who shared their data and expertise, worked diligently, and maintained good spirits throughout the workshop.

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Introduction to the Workshop

Sheryan Epperly

The objective of this workshop is to determine the feasibility of using sea turtle catch and survey methods to determine relative population abundance and population trends. The workshop will focus on methods using CPUE and transect data. Capture-recapture analyses will be presented to illustrate experimental design and data requirements.

The need for in-water work on sea turtles is self-evident. Sea turtles spend virtually all their time in the water. Historically most sea turtle research has focused on the brief emergences of nesting females, but this has been changing, particularly in the last decade. We have received encouragement at many times: the SEFSC Sea Turtle Habitat Plan of 1990 focused our research program on in-water activities; the National Academy of Sciences 1990 report recommended we initiate in-water activities to focus on life stages other than nesting; the recovery plans (1992-1995) identified the need for in-water research; and more recently the Turtle Expert Working Group (TEWG) recommended that in-water studies be continued and/or initiated. Numerous individuals have taken up the cry, including Karen Bjorndal in her talk at the Sea Turtle Symposium last year, where coincidentally I approached her about coordinating this workshop.

I would like to quote Peter Pritchard's article in the *Marine Turtle Newsletter* (January 1997), because it explains why in-water research is so important:

"It has been observed from time to time that a count of nesting turtles is a poor index to population size because it reflects recruitment a generation ago rather than gives information about the current vigor of the population. … Where many managers…have miscalculated is that they limited themselves to counting adults, becoming complacent if the numbers were good, and failing to protect eggs or take other means to ensure adequate recruitment into the population."

In some areas of the world we have a good measure of the adult nesting population, but actually know very little about the condition of the rest of the population. We may be experiencing recruitment failure of an entire series of cohorts, but we wouldn't know it until the cohorts became mature – unless we also were counting the younger animals in the water. If there was recruitment failure, and we failed to detect it until the animals matured, then the recovery of the population would be severely hampered given the long age to maturity of these species – it would be at least a generation time before we might begin to see the benefits of any management regime imposed.

From the standpoint of the population modeler, the stages prior to maturity are a big unknown for most species and populations. The TEWG was unable to determine the population size of the Kemp's ridley – one of the species that we know the most about – because at least one too many variables was unknown. If we had trend information we could have indexed the population model and could have narrowed the range of estimated population sizes.

The National Marine Fisheries Service is committed to recovering sea turtle populations. We recognize the need to determine the population size or trends in abundance and obtaining this information is one of our highest priorities. We are establishing Status Review Working Groups for each species (successors to the TEWG), and they will need this information in the future. We look forward to the forthcoming Workshop Report and will use it to guide our program in this regard. Your participation in this workshop is very much appreciated.

There are many reasons for us to be working in the water (such as studies of somatic growth, genetic structure, diet, and movement patterns); determining abundance is just one of those reasons. Much other information can be gained while collecting abundance data. However, we must realize that by focusing first on some of the other purposes, we may be undermining our ability to use the data collected in the study for abundance/trends purposes. Depending on the objectives of the study, this may be acceptable.

The objective of this workshop is to determine the feasibility of using sea turtle catch and survey methods to determine relative population abundance and population trends. It is also to train sea turtle biologists (us) in using population assessment analyses and in the experimental design needed to support these analyses. The variety of methods that we represent can be grouped into three categories: CPUE, transect, and capture-recapture.

I want to mention that this is not the first workshop for in-water researchers. Larry Ogren hosted one years ago. This workshop is building on the foundation laid at that workshop.

Review of Methods to Assess Abundance and Trends in Natural Populations

Brief Review of Evaluation of Catch per Unit Effort (CPE) Data

Mike Allen

The following is a summary of critical points to consider when evaluating data on catch per unit effort (CPE).

- CPE provides only an index of relative abundance, not an abundance estimate.

- CPE should increase with population density, but does not always do so, as shown in Figure 1.

- Hyperstability (Figure 2) results when:
  - the spatial scale is small
  - organisms are clustered
  - effort is concentrated where turtles are abundant

- Linear (proportional) relation occurs when fishing effort is random (Figure 3).
  - C = NEq, where C = catch, N = abundance, E = effort, q = capture efficiency

- Hyperdepletion can also occur (Figure 4)
  - stock appears depleted but is not
  - capture efficiency may decline due to gear avoidance

- Usually the relation between population abundance and CPE is not known.

- Often CPE-density relationships are highly variable when tested.

- In some cases, CPE is used as a qualitative index rather than a quantitative variable.

- In many situations, CPE cannot be used as an index of abundance due to sample biases, low catch rates, and highly variable catch rates.

- To avoid biases with CPE data, the following must be standardized:
  - sampling season
  - gear
  - factors influencing capture efficiency

- Sampling should be a random design if CPE is used quantitatively. Without some randomization, conclusions may not be general or apply to a broad region. Nonrandom samples would imply that conclusions only apply to the specific sites/area rather than the region as a whole.

- If standardized, nonrandom sampling can be useful for qualitative trends across years.

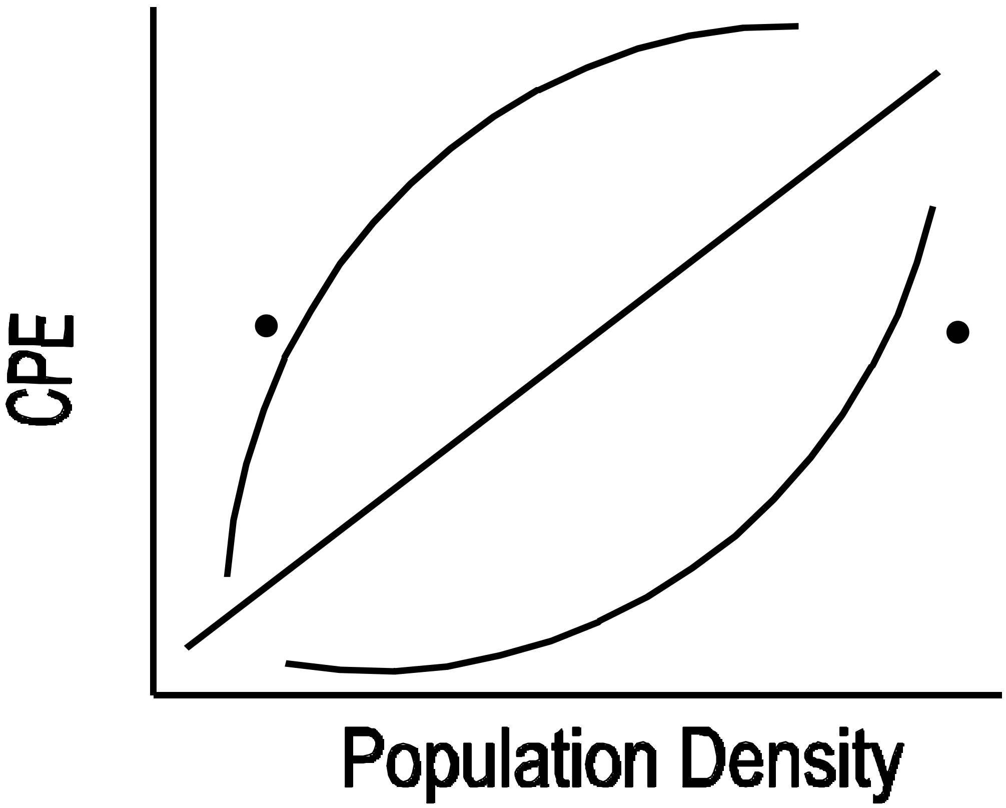

Figure 1. Possible relationships between CPE and population density, from Hilborn and Walters (1992).

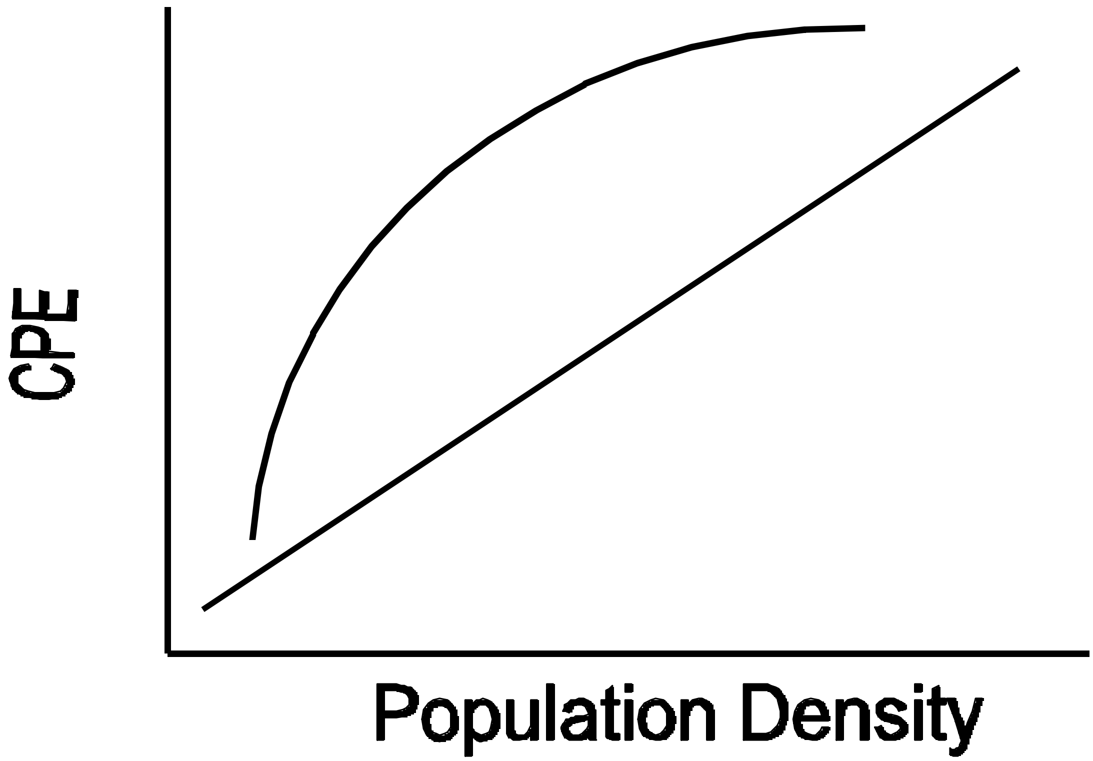

Figure 2. An example of hyperstability.

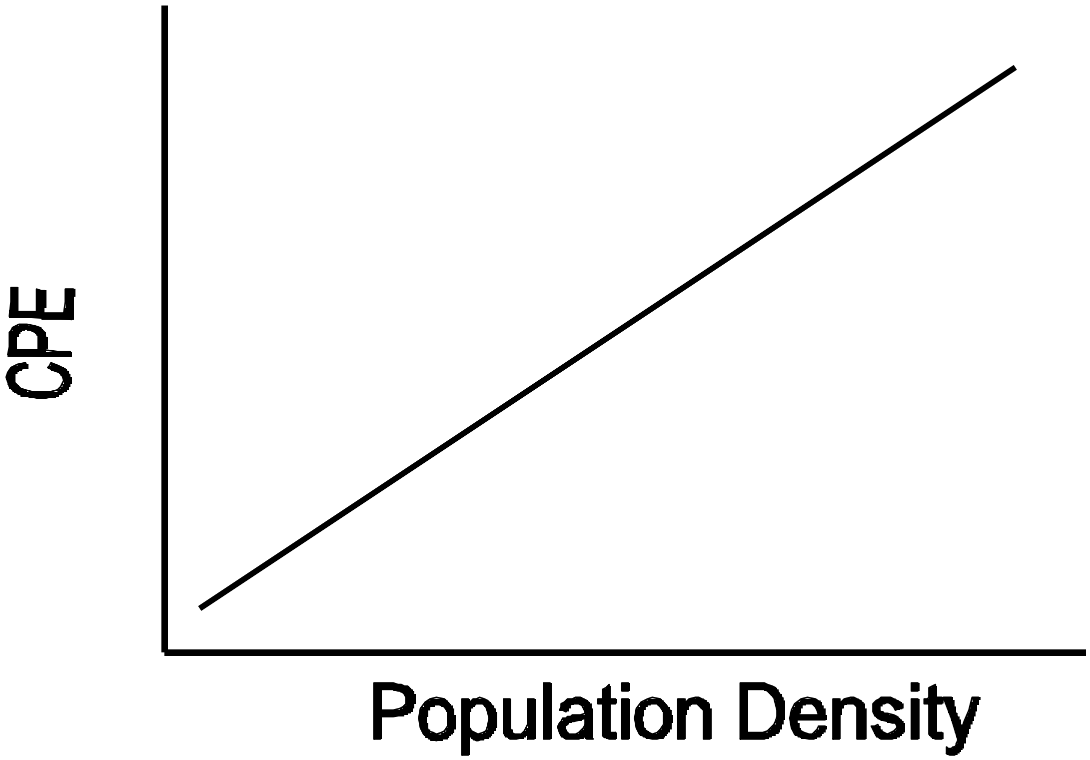

Figure 3. An example of a linear relationship.

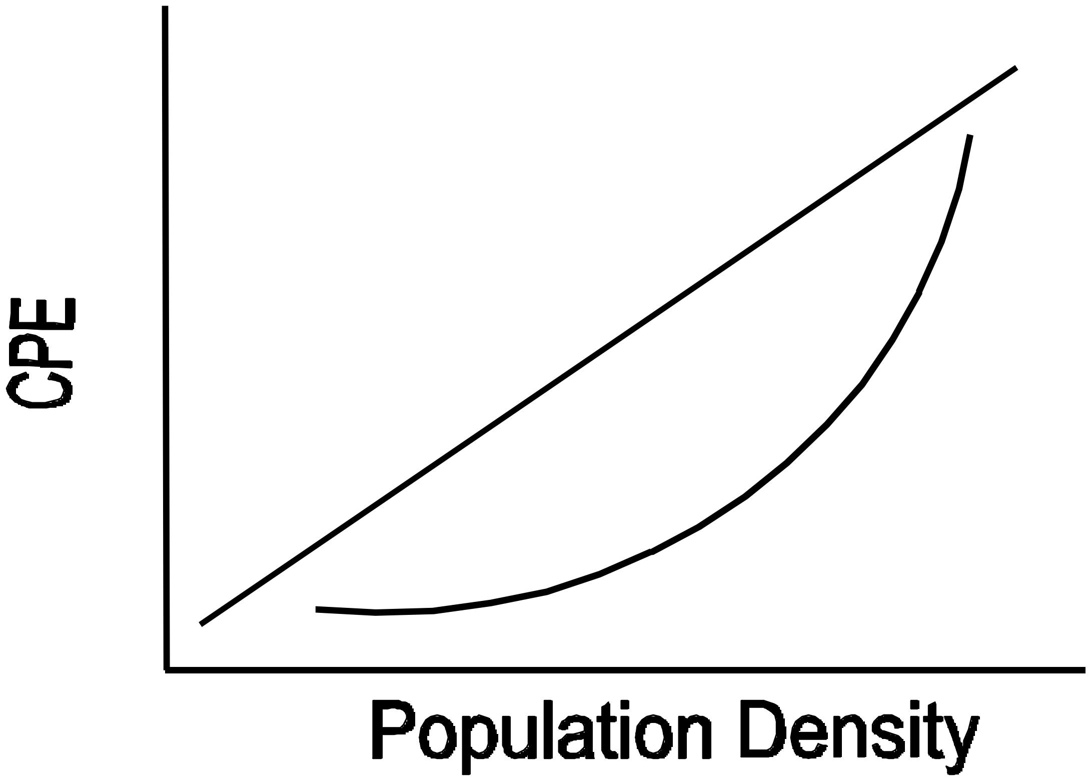

Figure 4. An example of hyperdepletion.

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Estimating Abundance with Transects

Tim Gerrodette

Transect sampling is the primary method of estimating abundance in wildlife studies. Transects may be as simple as swimming over a reef looking for turtles, or as sophisticated as a spatially stratified, aerial line-transect survey. The basic idea is that the number of turtles in a known area is counted, and density of turtles is computed by dividing the number counted by the area. Usually this is done many times (replicate transects). Finally, the number of turtles in a larger area is estimated from the random sample.

Basic Statistical Concepts

At the outset, it is useful to define some terms and concepts; for more details, a basic textbook on wildlife studies should be consulted, such as Caughley and Sinclair (1994) or Thompson et al. (1998). Seber (1982) is a more complete but more technical reference. *Sampling* is the process of gathering data in an organized way that will permit us to make inferences to a larger area or population. A *random* sample is one in which all members of the population have an equal chance of being chosen. Based on random samples, we make an *estimate* of abundance. All estimates are subject to *error*, which can be divided into two kinds. *Sampling error* arises because the world is not homogeneous, and therefore we do not observe, for example, the same number of turtles on each replicate transect. Sampling error is discussed in terms of the *precision* of the estimate. Common measures of precision are the *variance*, the *standard error*, the *coefficient of variation* (CV), and the *confidence interval*. Estimates of abundance should always be accompanied by (an estimate of) the precision of the estimate of abundance.

A second type of error arises when the sampling/estimation procedure does not “work right”, which leads to *bias* error. Bias is measured by *accuracy*, which means how close the sampling/estimation procedure would come to the true value on average. A bias may be either positive, meaning estimates tend to be too high, or negative, when estimates tend to be low. Of course we would like estimates to be both accurate and precise, but there may be a tradeoff between them. Precision can usually be improved by taking more samples, but accuracy cannot be improved in this way. That is, bias does not go away by collecting more data. Getting an unbiased estimate depends both on proper sampling and on using the proper statistical model for estimation.

Another useful distinction is between *absolute* abundance, which is the actual number of turtles in an area, and *relative* abundance (or an *index* of abundance), which is proportional to the actual number. For some purposes, such as determining the impact of a kill of turtles by fishing gear, the absolute abundance is needed. For other purposes, such as monitoring for a trend in population size, an index of abundance is sufficient. Besides being easier and less expensive, estimating relative abundance is often the only option, but it requires many assumptions that need to be carefully considered before accepting an index as valid.

Estimating from a Sample

Suppose we go along a line of length l, counting all turtles within a distance w of the line. Then we have counted all turtles in an area $a = 2wl$. If the number of turtles counted is n, then the density of turtles in our sampled area is $n \div a$. The basic assumption of estimating abundance from transects is that, because this is a random sample, the density of turtles in our sample is the same as the overall density of turtles in the study area. In other words, we assume that

$$\frac{n}{a} = \frac{N}{A},$$

where N is the true but unknown total population size and A is the total area. So the total population is estimated as

$$\hat{N} = \frac{n}{a} A$$

from the sampled quantities n and a and the known area A. Obviously it is important that the samples that produced n and a be chosen in a statistically valid way. It is *not* sufficient merely to select areas that are "representative." One way to obtain random samples is to select transect starting points and directions using a random number table. This may lead to some transects crossing each other, and thus some areas being sampled twice. Contrary to popular belief, there is nothing wrong with this (technically, it is called sampling with replacement). Another way to assure randomness is to conduct systematic sampling, which means covering the whole area with equally spaced transects (shown schematically in Fig. 1), with the starting point for the whole grid chosen in some random way.

The precision of the estimate of abundance is computed from the variability observed among the replicate transects. The way the transects are designed can have a strong effect on their variability. Consider Fig. 1, which shows a schematic study area with a gradient in density of turtles (the dots). The density of turtles increases from left to right in the study area. In Fig. 1A, the transects (the rectangles) have been placed parallel to the contours in density, so that there are large differences in the number of turtles encountered in the replicate transects. In Fig. 1B, the transects have been placed across the change in density, so that the number of turtles encountered in each transect does not vary as much. Because there is less variation among replicate transects, the variance of the estimate of abundance will be smaller. The placement of transects is therefore better in Fig. 1B than in 1A. Either sampling plan, however, will produce an unbiased estimate of abundance.

When it is known that turtle density varies strongly within an area, as in Fig. 1, or among separate areas, another option is to stratify the sampling – *i.e.*, to put more effort per unit area where there are more turtles. Generally, the rule is to stratify effort according to the square root of density on a per unit area basis. That is, if D_1 and D_2 are the densities of turtles in two habitats of areas A_1 and A_2, it is optimal to put a fraction

$A_1\sqrt{D_1}\Big/\left(A_1\sqrt{D_1}+A_2\sqrt{D_2}\right)$ of the transect effort in the first area. Of course, this requires knowing D_1 and D_2, which are the quantities we are trying to estimate, so stratification is only possible if we already have some estimates of density in each of the areas, either from previous studies or from preliminary sampling. These previous estimates of density should be reasonably good, because if effort is improperly stratified, it is possible to do more harm than good, in the sense that the stratified estimate of abundance will be less precise than the unstratified one. It should be emphasized that stratification affects only the precision of the estimate, not the accuracy. As long as the transects are properly placed within each stratum, the estimate of abundance will be unbiased, regardless of stratification.

The precision of the estimate of abundance $\hat{N}$ can be computed from the replicate transects in either of two ways. The first is to use the variance in density that is observed among the replicate samples to compute the variance of $\hat{N}$ as

$$\operatorname{var}(\hat{N}) = A^2 \operatorname{var}(n \div a).$$

Given the variance and assuming a normal distribution, a confidence interval can be constructed for $\hat{N}$. For example, the 95% confidence interval limits are

$$\hat{N} \pm 1.96\sqrt{\operatorname{var}(\hat{N})},$$

because 1.96 is the proper factor from a normal distribution to include 95%. The second way of computing the variance or confidence interval for $\hat{N}$ is to use a technique called the bootstrap (Efron and Tibshirani, 1993). This technique has come into widespread use with the availability of computers. The basic idea is to form a new set of data, called a bootstrap sample, from the original data by resampling, with replacement, the original set of transects. A new abundance estimate N_B is computed from this new set of data. The whole procedure is repeated a large number of times, say 1000 times, each time getting a slightly different estimate N_B. Then the variance of $\hat{N}$ can be calculated from the 1000 values of N_B, and the 95% confidence on $\hat{N}$ can be formed by sorting the 1000 values and finding the central 95% of them. The bootstrap does not rely on an assumption of a normal distribution, and in fact the point estimate $\hat{N}$ will usually not be in the middle of the bootstrap confidence interval. Use of the bootstrap to estimate confidence intervals is generally recommended because it relies on fewer assumptions.

Strip, Line and Point Transects

Transect sampling can be carried out in several ways. With strip transects, all turtles within a given distance of the line are counted, but turtles beyond that distance are not. The basic assumption is that the probability of detecting a turtle does not decrease from the center to the edge of the strip. This will not be true unless the width of the strip is quite narrow, much less than the farthest distance at which turtles can be “easily seen.” There is a strong tendency to overestimate the ability to detect turtles (or any animals) at a distance, and hence to make the strip too wide. This results in turtles being missed, and thus a negative bias in the estimate of abundance. If the strip is very narrow, we avoid this problem, but the survey is not very efficient because many of the turtles seen are beyond the strip boundary and are not counted. Furthermore, there is no way to check the assumption that there is equal detection probability across the whole width of the transect.

For these reasons, line transects have largely replaced strip transects. Line transects make use of all sightings and estimate the probability of detecting turtles at different distances from the trackline. With line transects, turtles within any distance of the line are counted. In addition, the perpendicular distance of each turtle or group of turtles from the line is measured. A quantity called the "effective strip width" is estimated from the data, based on how the number of animals detected declines with distance from the line. A typical detection curve is shown in Fig. 2. The effective strip width for these data is 2.1 km, but it can be seen that if we had to choose a strip width within which *all* animals would be detected (that is, within which the detection probability is not substantially below 1.0, which is the assumption of strip transects), the strip would have to be very narrow, in this case not more than 0.5 km (Fig. 2). Thus, it can be seen that line transects are more efficient than strip transects. The detection curve depends on the characteristics of the species (size, color, behavior), the way the animals are detected (height above water, boat or airplane), and also on the conditions at the time of the survey (sea state, for example).

If the objective is to estimate an index of turtle abundance rather than absolute abundance, many of these issues could be simplified. For example, the simple sighting rate (turtles/km) of all turtle sightings regardless of distance from the line could be used as an index of abundance without estimating an effective strip width. Implicitly this assumes that sighting conditions were the same at the times the transects were conducted. If the sea was very calm on the day the transects were conducted one year, and rough the next year, it would not be valid to use the sighting rate as an index of abundance between the two years, for example.

Both strip and line transects assume that turtles do not react to the observer before they are seen. This may be a problem for transects conducted from a boat or by swimming. If turtles react by moving out of the surveyed area before they are detected, this will create a negative bias in the estimated abundance. Point transects may help. With point transects, the observer remains at one spot instead of moving along a line, and the areas counted are circular or semicircular instead of rectangular. To avoid the problem of reaction, the observer must move to the observation spot and wait for some length of time until the animals have returned to their normal condition. The length of time for this to happen requires some testing. Turtles may be counted either within a fixed distance from a point, analogous to strip transects, or, preferably, all turtles regardless of distance are counted and the distance to each sighting is measured, analogous to line transects.

Sea turtles may spend a considerable amount of time below the surface of the water, and not be "available" to be detected by boat or airplane surveys. This is not a problem for estimating relative abundance, assuming the same fraction are submerged on each survey. On the other hand, if we are interested in absolute abundance, it is important to conduct additional research on diving behavior to estimate the fraction of turtles below the surface at any one time.

The standard reference for estimating abundance from transect sampling is Buckland et al. (1993). Excellent software is available for estimating abundance from strip, line or point transects, including making bootstrap estimates of variance. Both book and software are available for free on the internet at http://www.ruwpa.st-and.ac.uk/distance.

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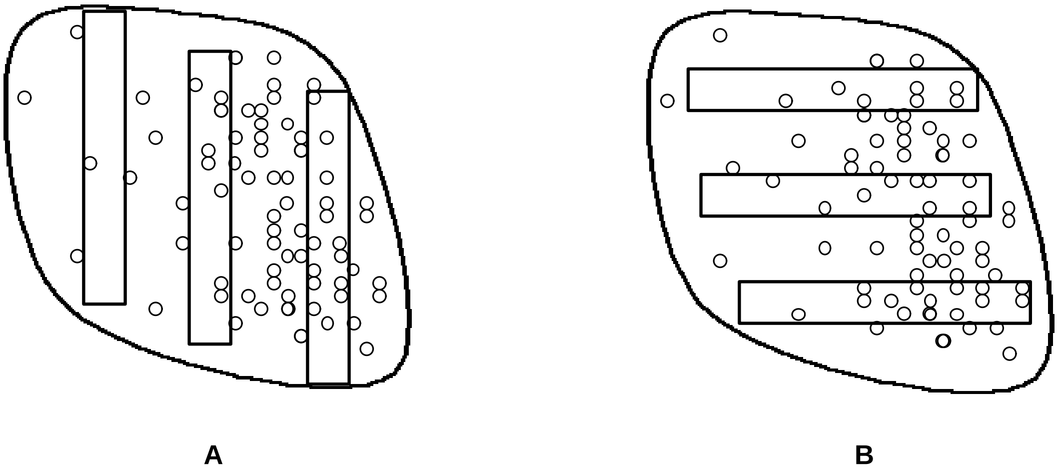

Fig. 1. Placement of transects when a gradient in density is present. The density of turtles (dots) increases from left to right in each figure. In A, the transects (rectangles) sample within similar densities and result in high variance. In B, the transects sample across density and produce estimates with lower variance. Both sampling plans, however, produce unbiased estimates of abundance.

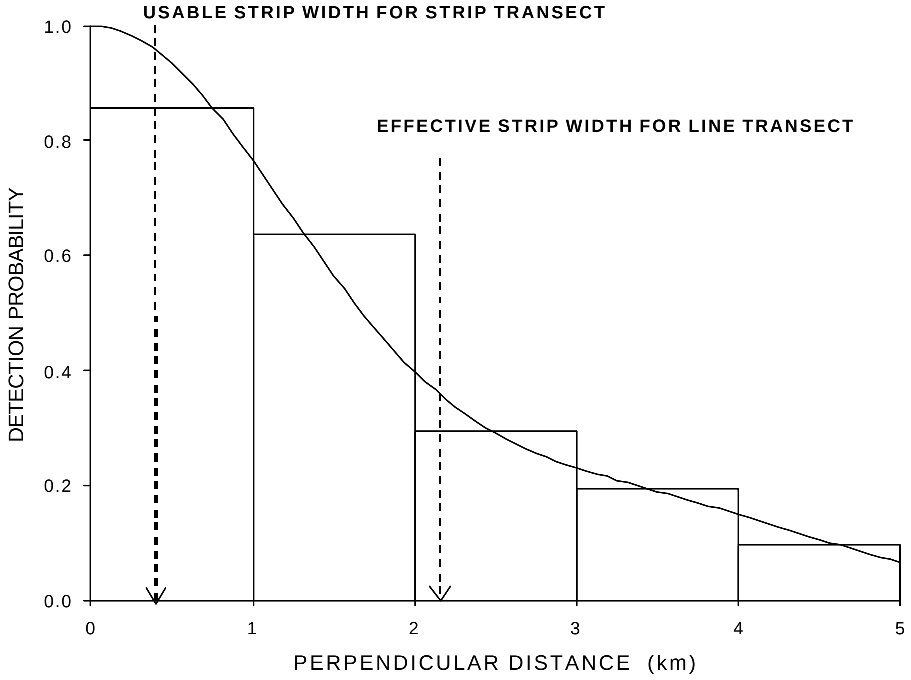

Fig. 2. A typical detection curve for line transects. The curve shows the probability of detection as a function of perpendicular distance from the transect line, estimated from the relative frequency of sightings at various perpendicular distances (the histogram). In this case, the probability of detection is shown to be 1.0 at distance 0 (*i.e.*, all turtles on the line would be seen), which would not be realistic for detecting turtles at sea. The effective strip width for these data is shown and contrasted with the usable strip width if a strip transect were conducted instead.

Capture-recapture Modeling of Sea Turtle Population Abundance

Milani Chaloupka

Purpose

The purpose of this presentation is to introduce some aspects of capture-mark-recapture (CMR) modeling that are applicable for the estimation of sea turtle population abundance in the foraging grounds. CMR includes mark-resight and sight-resight studies, but most sea turtle studies will most likely involve the actual capture, tagging and subsequent recapture to record individual capture histories.

Introduction

Sex- and age/stage-specific abundance are major components of the temporal and spatial dynamics of sea turtle populations and are essential for —

- population status and trend monitoring
- diagnosing demographic trends
- assessment of long-term population viability
- development of species recovery plans

There are 5 approaches that have been used to estimate sea turtle population abundance to support population status and trend monitoring —

- nesting beach census or sampling (Bjorndal et al. 1999)
- beach stranding counts (Epperly et al. 1996)
- CPUE estimation from trawl- (Butler et al. 1987) or logbook-surveys (Witzell 1998)
- line- (Epperly et al. 1995, Beavers & Ramsey 1998) or strip-transect (Marsh & Saalfeld 1989) sampling based density estimation
- capture-mark-recapture (CMR) type estimation (Le Gall et al. 1986, Chaloupka & Limpus in press)

Most population status and trend monitoring for sea turtles has been based on monitoring seasonal beach nesting activity. For instance —

- hawksbills (Bjorndal et al. 1993)
- loggerheads (Limpus & Reimer 1994)
- flatbacks (Parmenter & Limpus 1995)
- leatherbacks (Chan & Liew 1996)
- olive ridleys (Valverde et al. 1998)
- Kemp's ridleys (Márquez et al. 1999)
- green turtles (Bjorndal et al. 1999)

Monitoring beach nesting activity is important but provides insufficient information for population assessment for a number of reasons including —

- adult females skip breeding seasons and
- no information on demographic structure as immature, male and non-breeding females are not sampled.

Hence sea turtle abundance estimation depends on sampling the entire demographic structure of a population resident in the foraging grounds rather than being limited to sampling nesting females only. Here we focus on introducing CMR modeling for the estimation of sea turtle population abundance in the foraging grounds where all demographic components of a sea turtle population resident in the coastal benthic habitat can be sampled. The focus here is on population abundance estimation using samples of individually marked turtles. However, it is important to recognize that setting up and maintaining a reliable foraging ground CMR sampling study is a difficult logistic and expensive undertaking so that a long-term nesting beach study might well be the only feasible option in many circumstances.

Summary of Studies Using Marked Turtles

While sea turtles have been subject to a long history of exploitation (Parsons 1962, Frazier 1980) there have been very few abundance estimates for foraging ground populations. Some examples include Mendonca & Ehrhart (1982), Ross (1985), Le Gall et al. (1986), Butler et al. (1987) and Chaloupka & Limpus (in press). Mendonca & Ehrhart (1982) used closed population catch-effort (~ Otis et al. 1978 model M_b) and Schnabel-type estimators (~ Otis et al. 1978 model M_t) to assess immature green turtle and loggerhead abundance in an east-central Florida coastal lagoon.

Demographic closure means no population change between sampling occasions except for recapture behaviors. Open means that population changes between sampling occasions due to birth, death and dispersal processes in addition to recapture behaviors. Ross (1985) used a closed population Petersen estimator to assess green turtle abundance in foraging grounds off the Oman coast (northern Indian Ocean). An example of a simple 2-sample Petersen estimator would be as follows – say at sampling occasion 1 that 100 turtles were captured, tagged and released alive. At occasion 2 shortly after, 200 turtles were captured. The second sample comprised 50 tagged turtles from occasion 1 so that —

- ρ_2 = recapture probability at sampling occasion 2 = 50/200 = 0.25
- N_1 = No. marked/capture probability = 100/0.25 = 400

If a multiple sequential sampling occasion study was used rather than just 2 occasions, then the number of marked turtles at each sampling occasion could vary due to (1) new marking at each occasion and (2) birth, mortality and dispersal between sampling occasions. This more complicated open population study could be based on the Cormack-Jolly-Seber or CJS statistical modeling approach (Cormack 1989, 1993a&b; Lebreton et al. 1992) and various extensions (Brownie et al. 1993, Pradel et al. 1997, Kendall et al. 1997). While the CJS approach can be used to derive abundance estimates it has been used mainly to derive estimates of recapture, survival and recruitment probabilities. The more usual approach to abundance estimation using multiple sequential sampling is the Jolly-Seber or JS approach (Pollock et al. 1990, Schwarz & Arnason 1996, Schwarz & Seber 1999), but this approach is not as flexible as the CJS modeling approach for including covariates in estimating recapture probabilities that form the basis for abundance estimation. The CJS approach will be developed further here in this workshop for abundance estimation.

Le Gall et al. (1986) used an open population JS model (~ Pollock et al. 1990 Model A) assuming no skipped breeding seasons to estimate nesting female abundance at two Indian Ocean green turtle rookeries. Butler et al. (1987) used a 2-stage closed population modeling approach comprising —

- trawl survey sampling and a catch-effort or removal model (~M_b) to derive capture probabilities
- estimation of seasonal loggerhead abundance in Florida navigation channels over a 1 yr period

This capture-based abundance estimator is referred to here as a Manly-Parr type estimator and it forms the basis for the open population abundance estimator used in this workshop. More sophisticated closed population abundance estimation is possible using the approach of Huggins (1991) that also focuses on robust statistical modeling of individual recapture behaviors. The Huggins approach is applicable to closed populations while the CJS- or JS-related approaches are applicable to modeling demographically open populations. Chaloupka & Limpus (in press) used a 2-stage open population modeling approach comprising —

- rodeo-based CMR sampling to estimate sex- and stage-specific recapture probabilities for green and loggerhead turtles resident in Great Barrier Reef foraging grounds
- using those recapture probabilities to derive Manly-Parr type annual abundance estimates over an 8 yr period for the sGBR resident populations

The first stage in the 2-stage approach used by Chaloupka & Limpus (in press) was based on CJS statistical modeling of the survival, recapture and transient behavior of the sGBR green and loggerhead populations conditioned on annual sampling effort. The 2-stage modeling approach used by both Butler et al. (1987) and Chaloupka & Limpus (in press) focussed on deriving robust estimates of catchability or recapture probabilities, which then forms the basis for deriving robust estimates of sea turtle population abundance.

CJS Modeling Approach

The Cormack-Jolly-Seber (CJS) approach comprises simultaneous modeling of the temporal recapture (p) and survival (f) behavior of marked individuals (Lebreton et al. 1992), which is in contrast to the Jolly-Seber (JS) approach that addresses both marked and unmarked individuals. It must be stressed that the CJS approach addresses the MARKED individuals only so parameter estimates are applicable to the marked subgroup only but are presumed applicable to the entire population sampled in the CMR study. This is an important assumption of the CJS approach. The following programs are used to derive CJS estimates of recapture and survival probabilities that can then be used to derive population abundance estimates — GLIM (Cormack 1989, 1993a), SURGE (Lebreton et al. 1992) and MARK (White & Burnham 1999). Other programs and computer code are also used for CJS type estimation (see Lebreton et al. 1992, Catchpole et al. 1998). Programs JOLLY (Pollock et al. 1990) and POPAN (Schwarz & Arnason 1996) use the JS approach that focuses directly on abundance estimation while accounting for new recruits (births, immigrants) at each sampling occasion.

Why Use This Approach for Abundance Estimation?

Unlike the JS approach that focuses directly on abundance estimation, the CJS modeling approach in a CMR study can far more easily account for informative individual (e.g., sex, age) and/or sampling occasion covariates (e.g., sampling effort, sea surface temperature). This is then similar in intention to the closed population modeling approach of Huggins (1991). The CJS approach can also account for dispersal (Brownie et al. 1993) and transient (Pradel et al. 1997) behaviors that can have a major effect on estimation of recapture probabilities and population abundance. Hence the CJS approach provides a basis for robust estimation of survival and recapture probabilities conditioned on both informative individual and sampling occasion covariates. Usually recapture probabilities are addressed as a statistical nuisance parameter in the CJS model in order to derive robust survival probability estimates. However, the recapture behavior can be ecologically informative about the marked subgroup (Pugesek et al. 1995) and can be used to derive population abundance estimates (Manly & Parr 1968, Seber 1982, Loery et al. 1997, Schwarz & Seber 1999).

How to Use This Approach for Abundance Estimation

Some CJS background first. Assume an 8-occasion annual CMR study where the standard CJS parameter estimates are as follows in Table 1 (see Lebreton et al. 1992 for details):

	sampling occasion (*i*, 1…8)							
parameter	yr1	yr2	yr3	yr4	yr5	yr6	yr7	yr8
survival (ϕ_t)	ϕ_1	ϕ_2	ϕ_3	ϕ_4	ϕ_5	ϕ_6	ϕ_7	
recapture (ρ_t)		ρ_2	ρ_3	ρ_4	ρ_5	ρ_6	ρ_7	ρ_8
no. captured (n_t)		n_2	n_3	n_4	n_5	n_6	n_7	n_8

In other words, ϕ_1 occurs between sampling occasions 1 and 2 while ρ_2 occurs during sampling occasion 2 and so on. This is the full time-dependent CJS model with 14 parameters to be estimated although the parameters (ϕ_7 , ρ_8) are confounded with each other yielding only 12 useful parameter estimates. The parameter confounding is eliminated in most cases if reduced parameter model forms are used instead of the full CJS model (Cormack 1989, Lebreton et al. 1992) if for instance all the time-dependent survival probabilities were similar and hence indicative of constant mean annual survival.

Goodness-of-fit Tests and Model Selection Approach

The basis for using the CJS approach for abundance estimates is then founded on robust estimates of the recapture probabilities (ρ_i) and the number of turtles captured during each sampling occasion (n_i). However, the CJS assumptions must first of all be addressed prior to any parameter estimation. The major CJS model assumptions (Seber 1982, Cormack 1989, Lebreton et al. 1992) are as follows —

1. marked turtles have same capture probability (r)
2. marked turtles have same survival probability (f)
3. no tag loss
4. instantaneous sampling relative to inter-sample interval

Goodness-of-fit (GoF) tests are CRITICAL for robust CJS estimation and use of the CJS parameters for subsequent population abundance estimation. The CJS model fitting and GoF protocol appropriate here is outlined in the following six steps —

1. use RELEASE TEST2 + 3 to evaluate assumptions 1 and 2 above (Burnham et al. 1987, Lebreton et al. 1992), which can also be done from within program MARK (White & Burnham 1999) or program RELEASE with recent modifications for capture heterogeneity (see Pradel 1993)
2. assumptions 3 and 4 need to be assessed using other considerations and reliance on double tagging programs and use of non-corrosive metal alloy tags (see Chaloupka & Musick 1997)
3. if OK assess simpler model fits relative to the full parameter CJS model using AIC or AIC type variants (Burnham et al. 1995, Anderson et al. 1998) and/or loglikelihood ratio tests (Cormack 1993a, Lebreton et al. 1992)
4. if not OK then do not use CJS model for any parameter estimates (perhaps try some other approach)
5. if simpler model fits OK relative to full CJS model then use the simpler reduced or constrained parameter CJS model to derive survival and recapture probability estimates
6. use a parametric bootstrap GoF estimate of final referred model as implemented for instance in MARK (White & Burnham 1999) to assess overall preferred model GoF rather than just relative fit

TEST2 + 3 is used to evaluate the CJS assumptions as follows (see Burnham et al. 1987, Lebreton et al. 1992) – TEST2 helps evaluate assumption 1 regarding equal catchability while TEST3 helps evaluate assumption 2 regarding survival homogeneity. TEST2 failure is most likely due to temporary emigration behavior and/or recapture heterogeneity (Pradel 1993). TEST3 failure is most often due to age-specific rather than time-dependent survival or due to newly marked turtles displaying transient behavior (see Lebreton et al. 1992, Cormack 1993a, Loery et al. 1997). Age-specific models can then be fitted if appropriate using either SURGE (Lebreton et al. 1992) or MARK (White & Burnham 1999) while transience can be addressed using the Brownie-Robson type models (Pollock et al. 1990, Pradel et al. 1997, Loery et al. 1997). Use of these non-standard models specific to a particular data set is another reason why the use of CJS type modeling and robust derivation of recapture probabilities for abundance estimation is useful (see Loery et al. 1997) rather than using the JS approach.

Demographic Parameters

A good fitting full or reduced parameter CJS model can then be used to derive temporal or constrained estimates of several important demographic parameters including —

- apparent survival probabilities (ϕ_i)
- recapture probabilities (ρ_i)
- temporary emigration probabilities
- breeding and recruitment probabilities
- stage-specific transition probabilities
- proportion of transience/residence turtles

The most important parameters here for population abundance estimation purposes are the recapture probabilities conditioned on the individual and/or sampling occasion covariates. It is important to note here that recapture of a turtle during occasion $t+1$ that was captured during occasion t depends on the following 3 probabilities rather than just 2 as commonly occurs in CJS modeling — (1) probability of surviving from occasion t to occasion $t+1$, (2) probability of being present in the study area during occasion $t+1$ given that it is still alive, and (3) probability of being recaptured in the area during occasion $t+1$ given that it is alive and present in the area. The standard CJS estimates are suitable so long as there is no significant dispersal to other sites as the CJS model confounds mortality and emigration. If dispersal is important in the dynamics of the study population then a multi-site CMR study is needed. Robust CJS type recapture probability estimates can then be derived from such a study accounting for site-dependent dispersal behaviors using program MSSURVIV (Brownie et al. 1993).

Abundance Estimation Approach

Huggins (1991), Pradel et al. (1997) and Pugesek et al. (1995) have shown the value of focussing on recapture behavior where it is possible to model the effects of individual and/or sampling occasion covariates to derive recapture probability estimates. Loery et al. (1997) have then shown how it is possible to use such recapture probability estimates to derive robust population abundance estimates using the following approach. Assuming a good-fit CJS model to derive the time-dependent recapture probabilities (ρ_i) for the marked turtles, it is then possible to estimate abundance using a Manly-Parr (Manly & Parr 1968, Seber 1982, Loery et al. 1997) type maximum likelihood estimator as follows —

$N_i = (n_i / p_i)$ … (equation 1)

where n_i is the number of turtles captured in ith year or sampling occasion, N_i is the number of turtles (both residents and transients) in the population in ith year, and p_i is the estimated capture probability in *i*th year. An approximate 95% confidence interval can then be derived from —

$\{N_i \pm 1.96*se(N_i)\}$ … (equation 2)

where the conditional standard error $se(N_i) = [(n_i / p_i)^2 * (var(p)/(p)^2)]^{0.5}$ and var(*p*) is the estimated capture probability variance in *i*th year (Loery et al. 1997 for details).

It is also worthwhile to then consider using the variance components approach of Link & Nichols (1994) to distinguish between variability in abundance estimation due to either (1) sampling uncertainty or (2) ecologically relevant variation. This approach can be used in conjunction with either CJS population abundance estimation (Chaloupka et al. 1999 where sampling uncertainty accounted for $<$ 21% of the total variability in seasonal humpback whale abundance) or JS estimates (Link & Nichols 1994).

Some Sea Turtle CMR/CJS Study Examples

Manly-Parr type abundance estimates based on CJS recapture probabilities have been used recently to derive sex- and stage-specific abundance estimates for –

- green and loggerhead turtles resident in southern Great Barrier Reef waters (Fig. 1)
- green turtles resident in Union Creek (Bahamas, Fig. 2)

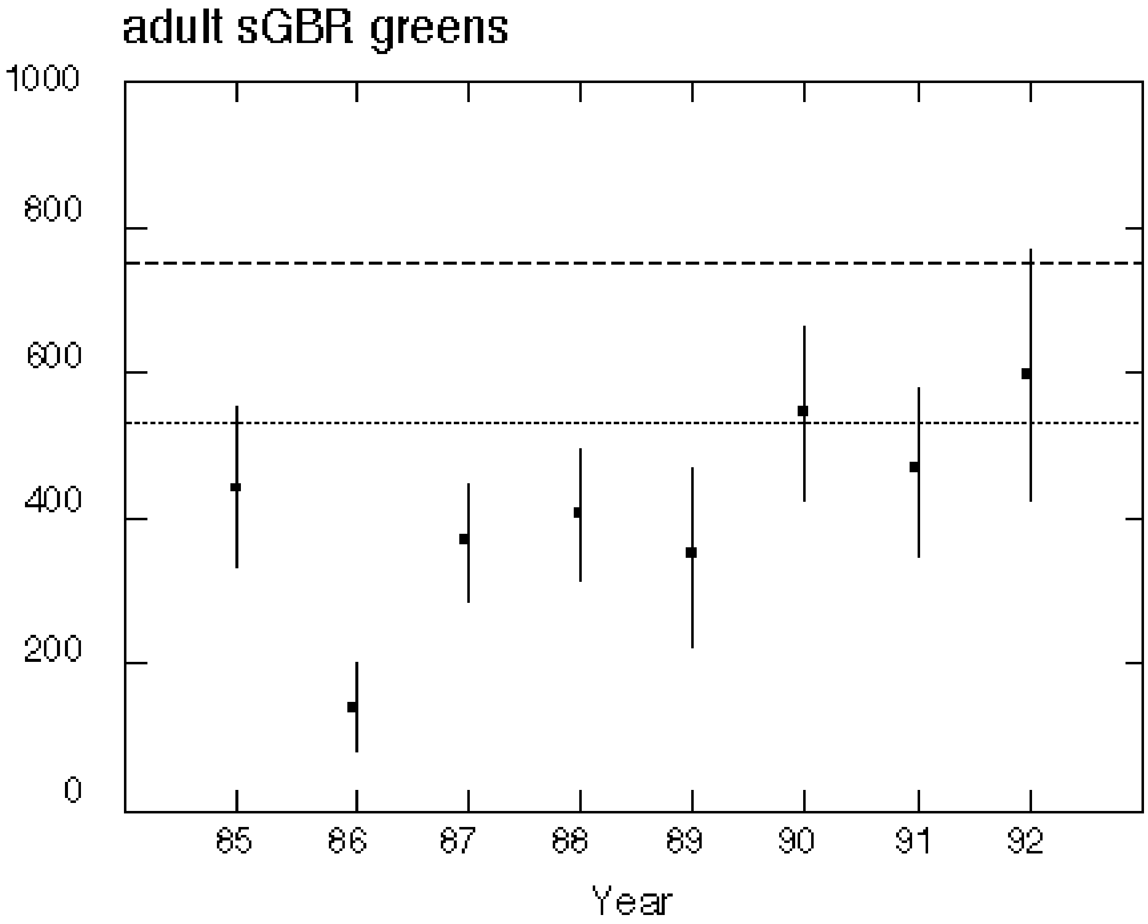

Figure 1 Annual Manly-Parr abundance estimates for green turtles resident in southern Great Barrier Reef waters between 1984 and 1992 (Chaloupka & Limpus in press). Sex-pooled mean stage-specific abundance estimates shown by solid square. Vertical bar = approximate 95% confidence interval. Dotted horizontal line = mean annual abundance (residents + transients) derived using a Poisson likelihood modeling approach (Cormack 1993a) for comparison with Manly-Parr type estimates. Dashed line = mean annual abundance using model M_{tbh} (Huggins 1991) accounting for individual capture heterogeneity, prior capture history, and individual and sampling occasion covariates.

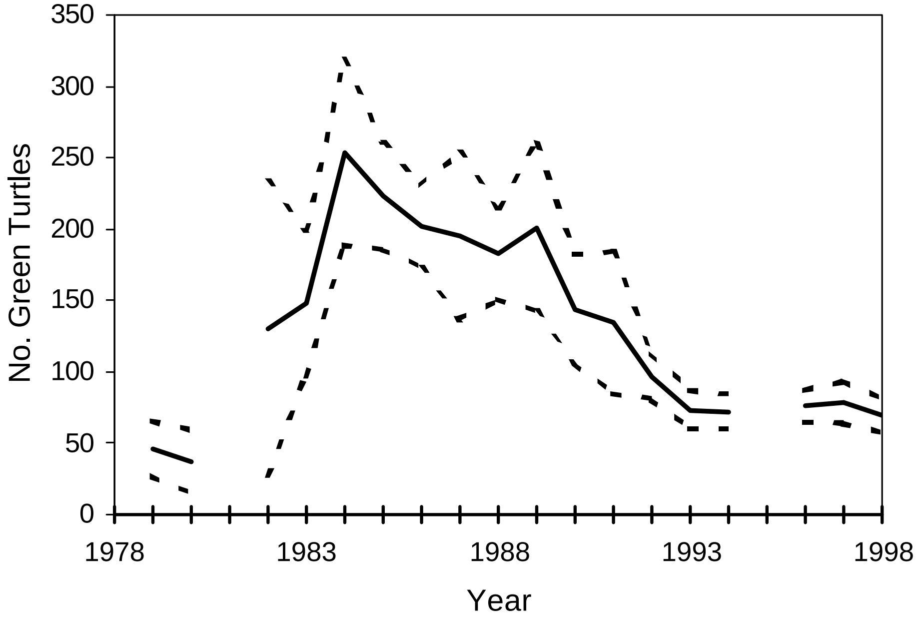

Figure 2 Annual Manly-Parr abundance estimates (solid curve) for immature green turtles resident in Inagua waters (southern Bahamas) between 1978 and 1999 derived using a CJS modeling approach accounting for potential transient behavior (Bjorndal & Bolten, University of Florida). Dashed curves show approximate 95% confidence bounds.

The population abundance series for these long-term study sites were discussed in detail at the workshop and show the utility of long-term abundance estimates for assessing the population status of these two sea turtle stocks. But abundance without estimates of say survivorship can be misleading. For instance — adult loggerhead abundance in sGBR waters has declined since the early 1980s, and this has been attributed to declining adult survival due to incidental mortality in otter trawl fisheries. But survival estimated using CJS models has not declined (Fig. 3), so the declining sGBR loggerhead stock must be due to something else such as incidental mortality in distance water longline fisheries for instance.

The important message here is that population abundance estimates of any form are important and necessary but not sufficient for diagnosing population trends. Robust estimates of other demographic parameters such as sex- and age/stage-specific survival probabilities are essential. Hence the more comprehensive nature of CJS modeling compared to say JS modeling as estimates of many important demographic parameters including abundance estimates can be derived using the same integrated approach.

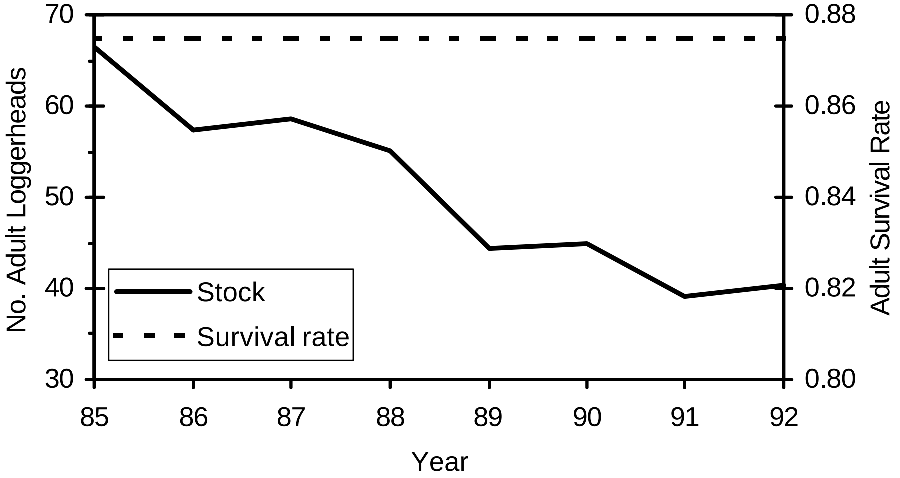

Figure 3 Manly-Parr type abundance estimates for sGBR loggerheads between 1985 and 1992 derived from CJS modeling approach (source: Chaloupka & Limpus in press)

Finally, there are many other issues that need to be considered in a thorough CMR study including consideration of the following –

- survey sampling design (single site vs multi-site)
- individual capture heterogeneity
- age, year, cohort effects
- CMR/CJS power analysis

These CJS issues are complex and were not addressed explicitly at this workshop.

Hands-on Demonstration

A brief hands-on demonstration was then presented at the workshop on how to derive CJS based Manly-Parr abundance estimates for an actual sea turtle population including issues related to the following —

- appropriate CMR data structure to use for CJS modeling
- use of programs MARK/RELEASE for GoF testing (TESTS2 + 3 and bootstrapping)
- use of program MARK for the CJS best fit model selection strategy
- use of program MARK for CJS parameter estimation (White & Burnham 1999)
- MS Excel to calculate Manly-Parr type estimates from the recapture probabilities

The demonstration used the study of Drs. Karen Bjorndal and Alan Bolten at Union Creek (Inagua) that focuses on immature green turtles (Fig. 2). The CJS modeling demonstration example addressed temporal and stage-specific demographic effects on both survival and recapture estimation as well as the possible transient behavior of the immature greens and what effect transience had on recapture probability estimation for this data set.

CMR data structure

The data set used in this demonstration comprised the annual capture-mark-recapture (CMR) history profiles for 670 immature green sea turtles resident in Inagua, southern Bahamas between 1978 and 1999. Details of the study site and CMR sampling procedures can be found in Bjorndal et al. (2000). The CMR sampling program enabled an annual capture-mark-recapture history profile to be recorded for each of the 670 individual immature green turtles recorded in the Inagua study area over the 22 year study period.

Each of the 670 CMR profiles recorded whether or not a particular green turtle was captured at least once during each of the 22 annual sampling occasions (1978-1999). For instance, the CMR profile {#12345: 0011010000001000000000} means a specific turtle was first captured, tagged uniquely (#12345) and released alive in the 3rd sampling occasion of the study (1980). It was then recaptured at least once in 1981 and again in 1983 and then again in 1990. The failure to recapture the turtle since 1990 does not necessarily mean that the turtle has died, it just simply has never been recaptured for whatever reason. The first "1" in each profile indicates initial marking and defines the capture cohort for each turtle. This CMR capture history form is known as "incomplete registration of survivors" because recapture probability < 1 and so survival estimation requires simultaneous modeling of survival and recapture probabilities (Pollock et al. 1990, Lebreton et al. 1992, Cormack 1989).

This was a single site sampling study. A multiple-site sampling study designed to estimate inter-site movement probabilities or dispersal (see Brownie et al. 1993) would have a CMR data structure that might look like {#12345: 00AB0A0000000000000000} for a particular individual, which means that turtle #12345 was first marked during 1980 but only in the SITE A. It was then recaptured during 1981 where it was recorded in SITE B. It was then recaptured again in 1983 but in SITE A. The zeros in the individual CMR profile mean the turtle was not recorded in any site in that sampling occasion. If the study comprised 4 sites then A if captured in SITE A, B if captured in SITE B, C if captured in SITE C, and so on.

Both the single sampling stratum or multiple-sampling strata CMR profiles outlined above provide suitable data file input for most CJS modeling programs such as MARK (White & Burnham 1999), MSSURVIV (Brownie et al. 1993), RDSURVIV (Kendall et al. 1997), TMSURVIV (Pradel et al. 1997), and RELEASE (Burnham et al. 1987) — file structure converters are available to convert this data structure into the form suitable for input to program SURGE (Lebreton et al. 1992) or the JS program JOLLY (Pollock et al. 1990) provided in the POPAN package (Schwarz & Arnason 1996). The data format needed for input to the GLIM code developed by Cormack (1989, 1993a) for CJS analysis requires a slightly different structure using the individual CMR profiles but accounting explicitly for the complete multidimensional contingency table format underlying the Poisson likelihood modeling approach used. It is relatively straightforward to use the GLIM directive TABULATE to create this structure to implement the Cormack CJS modeling approach.

The first 25 individual CMR profiles for the N = 670 demonstration data set (inagua.inp) are as follows for input into program MARK —

```
1110000000000000000000    1  0  ;
1110000000000000000000    1  0  ;
1100000000000000000000    1  0  ;
1100000000000000000000    1  0  ;
1010000000000000000000    1  0  ;
1010000000000000000000    1  0  ;
1000000000000000000000    1  0  ;
1000000000000000000000    1  0  ;
0110110000000000000000    1  0  ;
0100011100000000000000    1  0  ;
0100000000000000000000    1  0  ;
0100000000000000000000    1  0  ;
0100000000000000000000    1  0  ;
0100000000000000000000    1  0  ;
0100000000000000000000    1  0  ;
0100000000000000000000    1  0  ;
0010000000000000000000    1  0  ;
0000111010000000000000    1  0  ;
0000110100000000000000    1  0  ;
0000110000000000000000    1  0  ;
0000110000000000000000    1  0  ;
0000101110000000000000    1  0  ;
0000100101000000000000    1  0  ;
0000100100000000000000    1  0  ;
0000100010000000000000    1  0  ;
etc.
```

The first 22 columns recorded the unique CMR history profile for each turtle followed by 2 columns that code 2 distinct subgroups in the data set that could be sex or age but here reflect 2 size classes. Note the fixed column format. Columns 1 = 1978, column 2 = 1979 and so on. For instance, here all of the first 25 records are for the smaller size class code (1 0) where the larger size class was then coded as (0 1). This is simply dummy coding for group effects as for regression/ANOVA modeling without a constant term (see Draper & Smith 1981). Each data record in the input file ends with a semicolon.

This file is read into MARK using the RECAPTURES ONLY data type option. Refer to Cooch & White (1998) and White & Burnham (1999) for details.

Goodness-of-fit

The first task is to invoke RELEASE within MARK to evaluate model fit for the full parameter CJS model using TEST2 + 3 discussed above to review compliance with assumptions 1 and 2. The full or global CJS model comprises recapture and survival parameters for each relevant occasion and separately for each of the 2 sizeclass subgroups (see Table 1 above in **How to use this approach** section). Overall, Total TEST2 + 3 suggests reasonable compliance with standard CJS assumptions indicated by $P > 0.05$ —

```
Goodness of Fit Results (TEST 2 + TEST 3) by Group

                        Group  Chi-square   df   P-level
                        -----  ----------  ----  -------
                        small    81.6813    49    0.0023
                        large    28.9953    43    0.9496

                        Total   110.6767    92    0.0898
```

However, compliance is very good for the large size subgroup but very poor for the small size subgroup. Further examination of the following summaries for TEST2 and TEST3 separately shows that TEST2 compliance is satisfactory for both size subgroups but TEST3.SR compliance is poor for the small size subgroup. In other words, the global CJS model fits the large size subgroup well but the small size subgroup fails to comply with the assumption of homogeneous survival probabilities. This lack of compliance only with TEST3.SR rather than TEST2 or TEST3.SM for the small size subgroup is indicative of age-specific survival effects rather than time-dependent survival.

```
         Summary of TEST 3 (Goodness of fit) Results

Group  Component  Chi-square   df   P-level  Sufficient Data
-----  ---------  ----------  ----  -------  ---------------
Group 1 3.SR        50.7780    18    0.0001
Group 1 3.Sm        13.0846    14    0.5199

         Summary of TEST 2 (Goodness of fit) Results

Group  Component  Chi-square   df   P-level  Sufficient Data
-----  ---------  ----------  ----  -------  ---------------
Group 1 TEST 2      17.8187    17    0.4004
Group 2 TEST 2       6.2565    15    0.9751
```

There was no significant evidence for transient behavior for this data set as evaluated previously using TMSURVIV (Pradel et al. 1997). Despite lack of assumption compliance for the small size subgroup the overall model fit (TEST2 + 3) was satisfactory for the purposes of this demonstration so we proceed with reduced model fitting to determine the most parsimonious fit to the data set. The best fit model would then be considered adequate for estimation of the following parameters —

- survival probabilities for each subgroup
- recapture probabilities for each subgroup
- population abundance estimates for each subgroup

Model selection

A summary of the 16 CJS models fitted to the 670 CMR profiles is shown in Table 2. The best fit model compared to the global or full parameter CJS model (Model 3 below) was Model 1 comprising subgroup-specific and time-dependent survival probabilities but subgroup-independent time-dependent recapture probabilities. Model selection was based on the quasi-likelihood corrected form of the Akaike Information Criterion (AIC), which is used for rapid screening of a large number of models (Burnham et al. 1995, Anderson et al. 1998).

The GoF of the best fit model selected by QAICC relative to the global model was then also assessed in absolute terms using a parametric bootstrap approach implemented in MARK (see Cooch & White 1998). The bootstrap method enables an assessment of whether Model 1 fits the data set overall rather than just being the best fit compared to the model set summarized in Table 2. The bootstrap evaluation of Model 1 fit suggests that the data set was reasonably well fit despite apparent lack of fit determined by TEST3.SR. Fitting an age-specific CJS model to the small size subgroup would perhaps result in the best overall fit but this was not undertaken here.

Table 2 Summary of CJS model fits. QAIC = quasi-likelihood corrected form of the Akaike Information Criterion, nep = number of estimable parameters in model. Best model fit was Model 1 indicated by lowest QAIC value.

	survival		recapture			
model number	subgroup-specific	time-specific	subgroup-specific	time-specific	QAICC	nep
1	yes	yes	no	yes	2906.9	58
2	yes	no	no	yes	2918.6	23
3	yes	yes	yes	yes	2935.2	76
4	yes	no	yes	yes	2940.7	44
5	no	yes	no	yes	2954.8	39
6	no	no	no	yes	2964.5	22
7	no	yes	yes	yes	2972.2	60
8	no	no	yes	yes	2979.1	43
9	yes	yes	no	no	3079.8	43
10	yes	yes	yes	no	3081.8	44
11	no	yes	yes	no	3125.1	23
12	no	yes	no	no	3126.4	22
13	yes	no	no	no	3191.9	3
14	yes	no	yes	no	3193.7	4
15	no	no	yes	no	3233.4	3
16	no	no	no	no	3234.6	2

Parameter estimates for best fit model

The recapture probability maximum likelihood estimates are summarized in Table 3 (lines 43-63) for the best fit CJS Model 1 below derived using program MARK. Lines 1-42, which summarize the survival estimates, are not presented.

Table 3 CJS parameter estimates

```
                                                         95% Confidence Interval
 Parameter   Estimate       Standard Error      Lower           Upper
 ---------  --------------  --------------  --------------  --------------

Subgroup-independent time-dependent recapture probabilities

      43    0.5000001       0.1066003       0.3024316       0.6975686
      44    0.6289597       0.1956357       0.2468786       0.8976004
      45    0.0000000       0.0000000       0.0000000       0.0000000
      46    0.3752781       0.1511014       0.1451851       0.6799621
      47    0.4791746       0.0874680       0.3164162       0.6464775
      48    0.5641172       0.0740718       0.4176219       0.7002148
      49    0.6951099       0.0596915       0.5676028       0.7983736
      50    0.7335674       0.0539844       0.6157208       0.8255155
      51    0.3278619       0.0501029       0.2380325       0.4323557
      52    0.6833225       0.0615885       0.5526143       0.7903318
      53    0.4077812       0.0608290       0.2959125       0.5301005
      54    0.4798464       0.0648238       0.3567028       0.6054886
      55    0.3499974       0.0662463       0.2332965       0.4879245
      56    0.8265077       0.0696076       0.6478556       0.9250153
      57    0.7774546       0.0732584       0.6037238       0.8890213
      58    0.7935426       0.0688423       0.6278078       0.8975232
      59    0.0000000       0.0000000       0.0000000       0.0000000
      60    0.8571764       0.0653965       0.6780988       0.9447482
      61    0.7880192       0.0755739       0.6049874       0.9002282
      62    1.0000000       0.0000000       0.9999937       1.0000063
      63    0.0625638       0.8955358       0.0000000       1.0000000
```

Abundance estimates

The Manly-Parr type population abundance estimates derived from the recapture probability estimates listed above are summarized in Table 4. The abundance estimates were calculated using equations 1 and 2 above (**Abundance estimation approach**). The population abundance estimates for the Inagua green turtle stock are shown in Fig. 2 above indicating a significant decline in the stock since the mid-1980s. However, as cautioned above, population trends need to be evaluated in conjunction with estimates of other relevant demographic parameters.

Table 4 Summary of population abundance estimates for Inagua green turtles. ni = number of turtles captured in ith year, Ni = Manly-Parr population abundance estimate, Ni(se) = conditional standard error of Manly-Parr estimate, LCL and UCL = lower and upper 95% confidence bound of Manly-Parr abundance estimate, pi = best fit CJS model estimate of recapture probability in ith year, se = standard error of recapture probability estimate in ith year, var = variance estimate of recapture probability estimate in ith year. Missing rows show years with no sampling effort so recapture probability is zero.

year	ni	Ni	Ni(se)	LCL	UCL	pi	se(pi)	var(pi)
1978	22							
1979	23	46	9.81	27	65	0.5000	0.1066	0.01136
1980	23	37	11.37	14	59	0.6289	0.1956	0.03826
1981	0							
1982	49	131	52.57	28	234	0.3753	0.1511	0.02283
1983	74	148	25.95	97	199	0.4992	0.0874	0.00764
1984	143	254	33.30	188	319	0.5641	0.0741	0.00549
1985	155	223	19.15	185	261	0.6951	0.0597	0.00356
1986	148	202	14.82	173	231	0.7336	0.0539	0.00291
1987	64	195	29.82	137	254	0.3279	0.0501	0.00251
1988	125	183	16.49	151	215	0.6833	0.0616	0.00379
1989	82	201	29.98	142	260	0.4078	0.0608	0.00370
1990	69	144	19.42	106	182	0.4798	0.0648	0.00420
1991	47	134	25.41	85	184	0.3499	0.0662	0.00438
1992	80	97	8.15	81	113	0.8265	0.0696	0.00484
1993	57	73	6.90	60	87	0.7775	0.0732	0.00536
1994	57	72	6.23	60	84	0.7935	0.0688	0.00473
1995	0							
1996	65	76	5.79	64	87	0.8572	0.0654	0.00428
1997	62	79	7.55	64	93	0.7880	0.0756	0.00572
1998	69	69	5.97	57	81	1.0000	0.0865	0.00748

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Designing a Monitoring Program to Detect Trends

Tim Gerrodette and John Brandon

A fundamental question in many sea turtle and other wildlife studies is whether the number of animals in a particular area is increasing or decreasing. To answer this question, we may consider setting up a monitoring program with the objective of detecting a trend in population size over time. Alternatively, we may be interested in assessing whether data already being collected are sufficient to detect changes. In either case, we are interested in evaluating whether a particular program is suitable to detect changes in population size. In this spirit, the emphasis in this paper will not be on analysis of particular data to see if they show a trend; rather, the emphasis will be on the characteristics of the data necessary to be useful in detecting trends. In other words, we will be interested in the experimental design of a population monitoring program.

There are important aspects of a monitoring program that are not discussed here. In particular, it is assumed that: (1) turtle monitoring sites have been chosen in a statistically valid way that permits inference to a larger area of interest; (2) sampling methods appropriate for assessing the turtle population have been developed; (3) sampling is carried out in a quantified and consistent manner to produce both an estimate (such as mean turtle catch per hour per 100m of net) and a measure of the precision of the estimate (such as the coefficient of variation, CV, of the estimate); and (4) sufficient preliminary data have been collected that an estimate of the CV is possible.

What Is a Trend, and How Do We Measure It?

Trend refers to a general directional change in some quantity, such as the number of turtles in an area, over some period of time. The change must be in a consistent direction. If the number of turtles increased for five years and then decreased over the next five years, we would not call this a trend. A trend may or may not be linear, but even if not linear, it will have a large linear component, and it is usually adequate to model it as a linear trend. With biological populations, change is often proportional over time, which is to say, change is exponential. Such data can be made linear in time by taking the logarithm of abundance, and this usually has the additional benefit of making variances more nearly equal.

Trends may be assessed by a variety of statistical methods. The most common is linear regression, but nonlinear and nonparametric methods can also be used to estimate trends in data. We will focus on ordinary least-squares linear regression because it is a robust and widely used procedure. The main assumptions of linear regression are that the estimates be independent, have constant variance, and be normally distributed. In the context of this workshop, CPE and transect estimates will generally be independent in each year, but mark-recapture estimates are usually correlated. Homogeneity of variance needs to be evaluated on a case-by-case basis, since it may depend on behavior of the animals and the specifics of the sampling procedure. Normality also needs to be evaluated, but is usually satisfied because the metric is often a mean value. For example, suppose we set 20 nets each year to catch turtles, and thus have 20 catch rate values (e.g., turtles/hour/100m of net) each year. The 20 catch rate values may not be normally distributed, but their mean will be.

Detecting a Trend

Whether a trend is "real" or not is usually evaluated by means of a significance test. In this standard paradigm, we take as the null hypothesis that no change is occurring (the slope of the regression line through the data is zero), and test this against a specific alternative hypothesis that the population is increasing or decreasing at a particular rate (the slope has a particular value). A type 1 error occurs if the null hypothesis is true but is rejected; a type 2 error occurs if the alternative hypothesis is true but the null hypothesis is not rejected (Table 1). The probabilities of making type 1 and 2 errors are labeled α and β, respectively. In considering the adequacy of an estimate or index to track changes in population size over time, and hence to detect trends, we will be concerned with a concept known as statistical power. Power is the probability of correctly rejecting the null hypothesis, when in fact the alternative hypothesis is true (Table 1).

Table 1. Evaluation of null and alternative hypotheses by means of a significance test. The probability of making a type 1 error is α, and the probability of making a type 2 error is β. Statistical power is the probability of rejecting the null hypothesis when the alternative hypothesis is true.

Null hypothesis is	Null hypothesis true	Alternative hypothesis true
Accepted	Correct $P = 1 - \alpha$	Type 2 error $P = \beta$
Rejected	Type 1 error $P = \alpha$	Correct $P = 1-\beta =$ power

Since the objective of a monitoring plan is to detect changes, it is important to evaluate the power of any proposed monitoring plan. That is, we should estimate the probability that the proposed plan will be able to detect a trend. Although this may seem obvious, power is infrequently estimated (Peterman 1990). An effective monitoring plan will have high power of detecting a trend. We do not want to spend time and money on a monitoring program that has weak ability to detect changes. Furthermore, if we carry out a plan with such low power and then detect no change, we are on shaky ground concluding that no change is occurring. We need to evaluate the probability that the program could have detected a trend (i.e., had high power). If power is low, little weight can be given to the finding of no trend.

Fortunately, it is possible to estimate power given a few simple quantities. The power of a monitoring program to detect a trend depends on four main factors:

(1) The duration of the study. The more years the population is monitored the more likely it is that a trend will be detected, given that a trend is present.

(2) The precision of the procedure for assessing or indexing the population. The more precise the estimates are (lower variance, lower CV), the more likely it is that a trend will be detected. The precision will depend on many factors, including how the population of interest is dispersed, how the population is assessed (for example, by mark-recapture, transect, or catch-per-unit-effort), and how much effort is put into each assessment (for example, how many nets are set).

(3) The rate at which the population actually is changing. It is easier to detect high rates of change than low. Other things being equal, it is more likely that a 10%/year change in population size will be detected than a 2%/year change.

(4) The type 1 error rate. Type 1 and type 2 error rates are inversely related, so that as α increases, β decreases, and vice versa. Since power = 1- β, this means power increases when we are willing to accept a higher type 1 error rate (higher α). The type 1 error rate is usually chosen to be $\alpha = 0.05$, but this is a matter of choice.

Program TRENDS

Although it is intuitive how power qualitatively depends on these factors (for example, more precise samples have higher power), the quantitative relationship is complex (Gerrodette 1987, 1991). A computer program called TRENDS carries out the necessary calculations, assuming linear regression is used to assess trend (Gerrodette 1993). With this program, power can be computed for a given combination of duration, precision, rate of change, and α. Furthermore, the program allows the calculation of any of the five parameters, given the other four. For example, we can estimate the minimum detectable rate of change, when duration of study, precision, α, and power are specified.

TRENDS operates within the Microsoft Windows environment (a DOS version is also available). The user chooses which of the five parameters [duration of study, rate of change, precision (as measured by the CV of the estimate), significance level (α) and power (1-β)] is to be estimated and specifies values for the other four. The user can also specify a variety of other settings, such as whether change is positive or negative, whether the rate of change is expressed per time unit or overall, whether change is linear or exponential, and whether the significance test is to be conducted as a one-tailed test, a two-tailed test, or something in between, known as directional testing (Rice and Gaines 1994).

TRENDS can be used in planning a monitoring program to determine how many years are required, how much effort (sampling) is required each year, and what the probability is of detecting a change by the end of the study. This is called a prospective power analysis and is the most common use of the program. With caution, the program can also be used retrospectively (Thomas 1997), that is, to examine the probability of detecting a trend after the data have already been collected. It is important to be clear that TRENDS does *not* analyze a particular set of data to determine if a trend is indicated or not.

TRENDS is available over the internet at http://mmdshare.ucsd.edu/trends.html.

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Internet Web Sites for Software for Population Assessment

TRENDS:
http://mmdshare.ucsd.edu/trends.html

DISTANCE:
http://www.ruwpa.st-and.ac.uk/distance/

MARK (also the Colorado State University site with additional programs):
http://www.cnr.colostate.edu/ ~ gwhite/software.html

Patuxent Wildlife Research Center Software Archive:
http://www.mbr.nbs.gov/software.html

University of Manitoba Population Analysis Software Group:
http://www.cs.umanitoba.ca/ ~ popan/

A review of estimating animal abundance III (by Schwarz and Seber). This excellent review is available for downloading at the following site:
http://www.math.sfu.ca/mast/people/faculty/cschwarz/papers/1999/Review/

Web site for useful statistical analyses including power analysis and sample size:
http://home.clara.net/sisa

A very good web site for power analysis software:
http://sustain.forestry.ubc.ca/cacb/power/index.html

Individual Project Accounts

Assessment of Hawaiian Green Turtles Utilizing Coastal Foraging Pastures at Palaau, Molokai

George H. Balazs

Project Description

Long-term studies of green turtles in coastal marine habitats of the Hawaiian Islands have been underway since the late 1970's. The overall objective of this research is to obtain comprehensive conservation and management data relating to growth rates, health status, food sources, spacial distribution, foraging strategies, population trends, reproductive migrations, and underwater behaviors. Systematic monitoring for 27 consecutive seasons (1973-99) at the principal Hawaiian green turtle breeding site of French Frigate Shoals has documented a significant increase in nesting females. Palaau, centered at 21°06'N, 157°07'W along the south shore of Molokai, is one of many resident areas where green turtles aggregate in the Hawaiian Islands to feed and rest. Since 1982, turtles ranging from 35cm in straight carapace length to mature adults have been captured unharmed at this location as by-catch in a bull-pen net (Balazs et al., 1987, 1998). The basis of this fishing technique is similar to pound nets used on the U.S. Atlantic coast, and fish weirs constructed in the South Pacific, Philippines and Taiwan. The four panels of small-mesh net that make up the bull-pen are set in shallow water <2m deep to form a trap that fish and turtles enter, but are unable to exit. The bull-pen consisted of 975m of net until July 1988 when it was shortened to 610m.

Analyses

Catch per unit effort (number of turtles captured per kilometer/hour of net deployed), and the associated coefficient of variance, were calculated for the 18 years of bull-pen fishing (Table 1). The annual mean CPE ranged from 0.12 (1983) to 2.89 (1989), with an overall CPE of 0.97 turtles/km·hr and a CV of 0.68. An analysis of variance using the SAS system detected a significant increase in CPE over the 1982-99 study period (Figure 1).

TRENDS software used for power analyses indicated that a minimum rate of annual change of 6% in the Palaau green turtle aggregation could be detected using the bull-pen methodology employed for 18 years. A minimum of 11 years would be needed to detect an annual change of 25%.

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Balazs, G.H., W. Puleloa, E. Medeiros, S.K.K. Murakawa, and D.M. Ellis. 1998. Growth rates and incidence of fibropapillomatosis in Hawaiian green turtles utilizing coastal foraging pastures at Palaau, Molokai. *In* S.P. Epperly and J. Braun (comps.), Proceedings of the Seventeenth Annual Symposium on Sea Turtle Biology and Conservation, p. 130-132. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-415.

Balazs, G.H., R.G. Forsyth, and A.K H. Kam. 1987. Preliminary assessment of habitat utilization by Hawaiian green turtles in their resident foraging pastures. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-71, 107 p.

Table 1. Annual CPE Data for 18 Years of Bull-Pen Fishing Involving 74 Days that Resulted in the Capture of 1,685 Turtles

Year	XCPE	STD	STDERR	N	CV	MIN	MAX
1982	0.26	0.27	0.12	5	107.37	0.04	0.68
1983	0.12	0.13	0.06	4	102.77	0.02	0.30
1984	0.16	0.14	0.04	11	84.48	0.04	0.43
1985	0.51	0.43	0.12	14	85.35	0.04	1.62
1986	0.46	0.39	0.22	3	85.07	0.04	0.81
1987	0.43	0.30	0.17	3	70.00	0.13	0.73
1988	2.14	1.18	0.84	2	55.44	1.30	2.97
1989	2.89	1.47	1.04	2	51.05	1.84	3.93
1990	1.14	0.73	0.32	5	63.80	0.07	1.87
1991	0.90	0.58	0.29	4	64.12	0.14	1.54
1992	1.30	0.97	0.56	3	74.94	0.20	2.07
1993	1.16	0.09	0.05	3	7.84	1.07	1.25
1994	0.79	0.05	0.03	2	6.15	0.75	0.82
1995	0.92	0.72	0.51	2	78.57	0.41	1.43
1996	0.91	0.60	0.30	4	66.07	0.34	1.50
1997	0.98	0.16	0.11	2	16.44	0.87	1.09
1998	1.16	0.10	0.07	2	8.32	1.09	1.23
1999	1.26	0.58	0.33	3	46.39	0.68	1.84

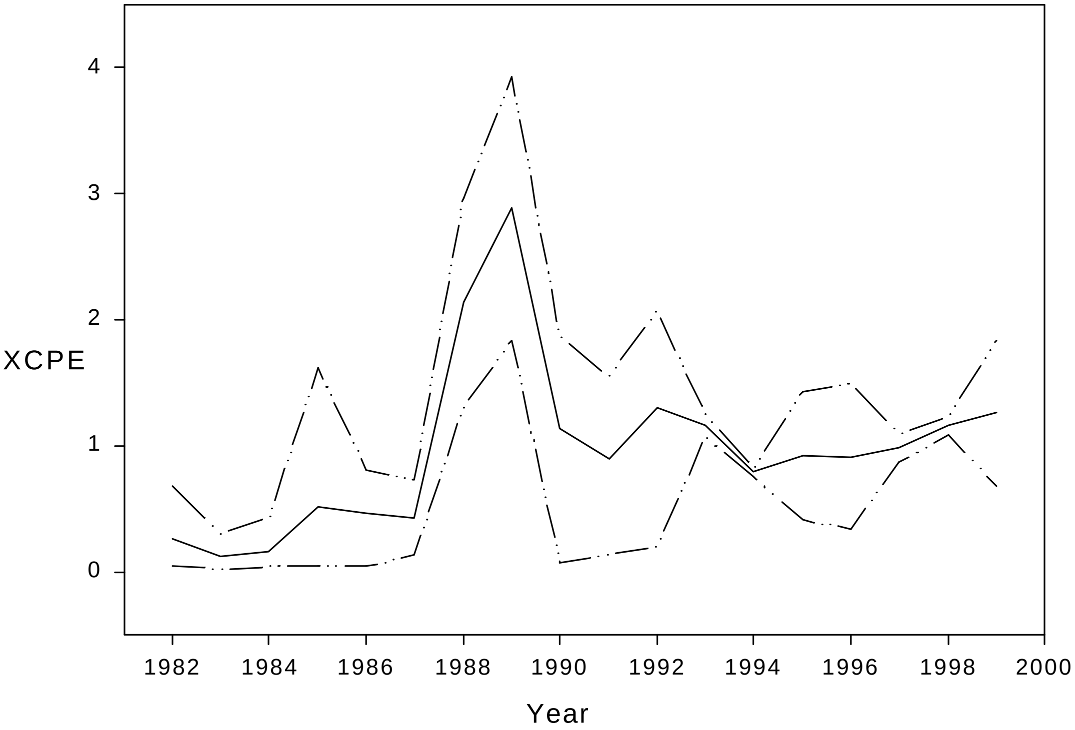

Figure 1. A significant increase has occurred in catch per unit effort of green turtles

Hawksbill Turtles at Mona Island, Puerto Rico

Carlos E. Diez

Project Description

In-water surveys were conducted at Mona Island from 1992 through 1999, and study sites varied from 2 to 7 areas. However, analyses were only conducted for three of the study sites for 1994 through 1999 because of small sample size in the other study sites and years. Turtles were captured by hand and others were sighted at the same time. Catch per unit effort was calculated by adding turtles captured and turtles sighted per total time spent in the water. Surveys were conducted once a year during a period of approximately 30 days. There were three observers in the water and one in the boat.

Analyses

CPE and CVs

Mean CPE and coefficient of variance were calculated for each study site and year (Table 1). From 1994 through 1999, all but one of the CVs were lower than 100, with a range from 40.07 to 101.06. Our mean CPE ranged from 2.26 to 9.35 turtles per hour. Univariate analyses of CPE per year and site indicated a normal distribution of the data in most cases. Further analyses in gear type have to be considered for calculating the CPE.

Power Analyses

Using TRENDS software, we tested whether we could detect a 10% change in the mean of our population at all sites and all years with an alpha level of 0.5. The power value was 0.547 with a CV of 0.28. All sites and all years were considered in this analysis. If we had limited the analysis to those years and study sites reported in Table 1, the CV and minimum sample size might have been reduced.

Table 1. CPE and CV statistics for the hawksbill project at Mona Island, Puerto Rico.

Year	Study site	Mean CPE	STD	N	CV
1994	Carabinero	3.39	3.43	38	101.058
	Monito	9.35	5.8	10	62.121
	Norte	5.12	2.48	28	48.503
1995	Carabinero	2.61	1.81	50	69.346
	Monito	7.84	3.19	19	40.661
	Norte	5.14	3.53	25	68.77
1996	Carabinero	2.72	1.61	32	59.111
	Monito	6.11	2.92	9	47.69
	Norte	4.3	1.88	17	44.197
1997	Carabinero	2.71	1.99	25	73.527
	Monito	8.89	4.24	10	47.653
	Norte	5.98	3.52	15	58.909
1998	Carabinero	2.32	1.38	28	59.55
	Monito	6.05	3.7	11	61.744
	Norte	3.5	1.4	5	40.078
1999	Carabinero	2.26	1.24	16	54.973
	Monito	8.34	5.58	10	66.893
	Norte	3.61	1.58	12	43.981

UCF Marine Turtle Research – Indian River Lagoon and Near-shore Worm Reefs

Llew Ehrhart and Bill Redfoot

Project Description

CPE data were derived from studies of two aggregations of juvenile green turtles and one aggregation of subadult loggerheads on the east central Florida coast. One of the green turtle groups and the loggerhead group reside in the Indian River Lagoon, a shallow inshore estuary where water depth is approximately 2 m and where there is little to no movement of water other than wind-driven movement at the surface. In most cases 460 m (12' deep) of large mesh tangle net (16" stretch, 8" on the bar) are set for varying periods of time (weather dependent, up to ca. 8 hrs). Both ends of the 3/8" polypropelene topline are attached to 13 lb Danforth-type anchors by 5' lengths of 5/16" chain and 100' of nylon line. The bottom line is composed of No. 30 continuous-lead-core line. "Bullet-shaped" styrofoam floats are attached at 30' intervals. The net is tended continuously by elevation of the topline from the bow of a boat.

The other green turtle aggregation resides over nearshore worm-rock reefs in the Atlantic. In this case the same 12' net is hung out fully in 10-12' of water. This is a high-energy shoreline and there is so much surge and flow of water that tending the net from a boat is ineffective. At least two swimmers snorkel the topline continuously and an adequate number of "helpers" are available to get in the water when multiple captures occur or a primary swimmer needs assistance. Far less net (220 m) is soaked in this situation, and the duration is typically 20 min to 3 hrs. Data gathered over 10 years in the lagoon and 11 years over the nearshore reef were presented and analyzed at the Population Assessment Workshop.

Analysis

We looked at ten consecutive years of capture rate data (CPUE) for both loggerheads and green turtles from the Indian River Lagoon and eleven consecutive years of capture rate data for green turtles from the near-shore worm-rock reefs off of Indian River County.

Abundance

Study Area	Species	Mean CPUE	Std. Dev.	N	CV	Min	Max
Lagoon	Green	2.28	3.80	251	167.0	0	34.78
Lagoon	Loggerhead	0.66	0.86	251	128.9	0	4.73
Reef	Green	14.15	24.47	190	172.95	0	147.42

CVs for TRENDS

Study Area	Species	CV
Lagoon	Green	0.11
Lagoon	Loggerhead	0.08
Reef	Green	0.13

North Carolina Pound Net Sampling

Sheryan Epperly

Project Description

The Pamlico-Albemarle Estuarine Complex is the largest estuarine system in the southeast U.S. and the second largest in the U.S. Three species of sea turtles (loggerhead, green turtle, and Kemp's ridley) utilize the sounds and tributaries as developmental habitat. The open waters of Pamlico and Core Sounds alone total over 4700 km^2. During their emigration in fall and early winter, the turtles are vulnerable to capture in pound nets set behind the barrier islands. Pound nets, as set in North Carolina, are a passive gear that allow turtles to feed and to surface to breathe.

Pound net fishermen of Pamlico and Core Sounds, N.C., were randomly sampled for 13 weeks each fall (mid Sept. - mid December) 1995-1997. Each week up to 12 trips were sampled. Soak time data (the number of days since the net was last fished) were recorded along with species-specific catch information. Approximately 1000 pound nets were sampled each year, and 100-150 turtles were captured each year. Aerial surveys were flown weekly to enumerate the total number of pound nets (up to 900/week were detected).

Analyses

Catch data were analyzed with parametric and non-parametric statistics.

Parametric statistics

Catch data, unadjusted for effort, for weeks 3-10 are reported in Table 1. Annual means of log10 transformations of the catch data were used in a regression analysis to determine CVs for trend analysis. For each of the 3 species, CVs were very large, due to the large number of pound nets sampled in which no turtles were captured. Trend analysis was not possible due to the very high variability.

Non-parametric statistics

Sample catch data were bootstrapped to derive normally distributed estimates of mean CPUE (Table 1). Log10 transformations of the annual mean CPUE values were used in a regression analysis to determine CVs for trend analysis. Trend analysis indicated that detectable rates of change over the 3-year period ranged from 0.076 to 1.518, depending on species. Estimates of the number of years to detect a rate of change of 0.50 ranged from <3 to 6 years.

This example clearly indicates the value of using non-parametric statistical models in the analysis of datasets with a large number of zero catches, which might be typical of random sampling for sea turtles. Also, trend analysis, based on only 3 years of data, are far too few to evaluate process error. Data from more years are needed to evaluate this method.

Table 1. Results of analyses. CPUE = catch/effort; catch = number turtles in a pound net; effort = soak time (days); N = number of pound nets fished.

	Loggerhead	Green Turtle	Kemp's ridley
Parametric Estimates of Abundance	Catch (std error), weeks 3-10 only, all 3 strata combined no adjustment for total number of pound nets fished in each strata		
1995 (N=750)	0.1013 (0.0141)	0.0480 (0.0080)	0.0013 (0.0013)
1996 (N=714)	0.1008 (0.0121)	0.0420 (0.0089)	0.0056 (0.0028)
1997 (N=792)	0.1540 (0.0176)	0.0290 (0.0060)	0.0114 (0.0038)
all years (N=2256)	0.1197 (0.0087)	0.0395 (0.0044)	0.0062 (0.0017)
Model:	Mean $\log_{10}$(Catch+1) = year		
Mean mean log10 Catch+1	0.0002	1.2234 E-7	3.3644 E-8
error mean square	0.0329	0.0116	0.0007
Bootstrapped Estimates of Abundance	Mean CPUE (95% confidence intervals), all 13 weeks, adjusted for total effort (pound-days fished) in strata		
1995	0.1025 (0.0804-0.1261)	0.0405 (0.0295-0.0521)	0.0014 (0.0-0.0040)
1996	0.0822 (0.0640-0.1020)	0.0315 (0.0188-0.0471)	0.0033 (0.0001-0.0078)
1997	0.1179 (0.0976-0.1405)	0.0216 (0.0136-0.0315)	0.0090 (0.0065-0.0153)
Model:	$\log_{10}$(mean CPUE) = year		
Mean log10 mean CPUE	0.3017	0.1171	0.0192
error mean square	0.00255	5.135 E-6	0.00004
C.V. $_{residuals}$	0.1674	0.0193	0.3294
Power Analysis	alpha = 0.2, power = 0.9, rate of growth = exponential, rate of growth sign = negative, 2-tailed test		
rate of change that can be detected when duration=3 years	0.687	0.076	1.518
number of years to detect a rate of change of 0.50 (couldn't select overall)	4	<3	6

NMFS Trawler Observer Data (1973-1982)

Terry Henwood

Project Description

The NMFS observer database consisted of four separate projects which were merged to estimate sea turtle incidental catch and mortality aboard shrimp trawlers. The purpose of these projects varied from estimating bycatch of fish, to testing of excluder gear, to directed observations of turtle catch and mortality. When merged, the file contained information on length of tows, location, known net types and sizes, and documented turtle takes.

This dataset would not be expected to be useful as a repeatable survey to document changes in populations over time. In fact, turtles at that time had been subject to years of shrimping related mortality, and catch of turtles was considered to be a rare event. Using trawlers to collect CPUE information was never considered by NMFS due to the low precision associated with CPUE estimates and the high costs associated with collecting this type of information.

In addition to the low turtle catch rates observed on shrimp vessels at that time, the data could not be used to address population distributions and abundance. The turtle catches represented catches by shrimp vessels on the shrimping grounds. Nothing could be said about turtle distribution or abundance in areas where shrimping was not occurring.

Analyses

The data were analyzed according to methods suggested in the workshop for loggerhead and Kemp's ridley turtles using the years 1978-1981. With all data for the Gulf of Mexico and the South Atlantic combined, the CV for loggerheads was 6.4366 and for Kemp's ridley was 41.819. The conclusion was that trawl data with catch rates expected at that time would not provide useful information on trends.

Population Assessment of Kemp's Ridley Sea Turtles in the Northwest Gulf of Mexico

Andre M. Landry, Jr.

Project Description

Entanglement netting operations have been conducted on a monthly basis from May through October to assess population dynamics of Kemp's ridleys at Sabine Pass, Calcasieu Pass and/or Mermentau River since 1992. These operations incorporate 91.4-m long entanglement nets set at 1 to 4 fixed stations adjacent to beachfront and jettied pass habitats. All stations exhibit water depths between 1.3 and 3 m. One to two nets are set at each station for 6-10 hrs per day. These surveys typically achieve a minimum of 5 netting days/month/study site. All nets are checked for sea turtles and associated bycatch every 30 minutes or upon visual/audible cues of said captures. Ridleys are enumerated, measured for straight/curved carapace length, and weighed. Prior to release, all turtles are tagged with Inconel tags on the trailing edge of each foreflipper and PIT tagged in the dorsal musculature of the left foreflipper.

Project Objectives

1) Characterize spatial and temporal abundance, size composition and sex ratio of Kemp's ridleys across nearshore habitats on the upper Texas and southwest Louisiana coasts.
2) Assess the role beachfront and tidal passes play as developmental and reproductive conditioning habitat for Kemp's ridleys.

Data Analysis

This analysis was limited to capture totals and netting effort (expressed as kilometer-hours) for only summer months (June-August) at Sabine Pass, the only season and study site characterized in a similar manner across years. These data were used to generate CPUE (catch per unit effort) by year for the period 1992-1999. All data were log transformed. A SAS program was used to analyze CPUE across years, while the program TRENDS was employed in a power analysis for trend detection.

Results

Capture totals, mean catch and associated statistical parameters for all ridleys taken in summer surveys from 1992-1999 are shown in Table 1.

Analysis of variance on CPUE failed to yield significant difference across years for untransformed data but the reverse for transformed data. Additional inspection of these data yielded the following:

1) mean annual CPUE ranged from 0.1 to 1.7 ridleys/km-hr, with overall CPUE equal to 0.9 ridleys/km-hr; years of highest CPUE were those also exhibiting greatest standard deviation;

2) only one of 8 years yielded data which were normally distributed; the year in question was one where sampling effort and capture totals were the lowest recorded;
3) 5 of 8 years yielded unacceptable coefficients of variation (acceptable being defined as $\leq$100); these ranged from 65.996 to 282.866 and averaged 172.172;
4) paired tests between years (using transformed CPUE data found significant by a Kruskal-Wallis Test) yielded significant differences for only years of highest and lowest CPUE;
5) variability in annual sampling effort (kilometer-hrs) was highly significant;
6) years with highest sampling effort yielded lowest CPUE and greatest variability;
7) CV calculated around the mean and residual was 0.2 and 0.39, respectively; the latter CV was used in the power analysis.

Discussion

The results summarized above are influenced by the fact that, in an attempt to analyze a data set generated in the same manner (summer season only) across years, months with higher CPUE (i.e., spring and fall) were not included. One probable artifact of this omission is that overall annual CPUE will be higher than that reported for the analysis above. Another possible artifact of including months with larger capture success would be impact on overall variability as well as coefficient of variation. Consequently, analyses inclusive of all months, regardless of sampling inconsistencies across years, is a prerequisite to better understanding of Kemp's ridley data from the northwestern Gulf of Mexico.

Table 1. Capture totals, mean catch and associated statistical parameters for all ridleys taken in summer surveys from 1992-1999

Year	**N**	**Mean**	**Std Dev**	**STDERR**	**CV**	**CVX**
1992	6	1.51	0.9933415	0.40553	65.996	0.26943
1993	43	0.90	1.7110068	0.26093	188.928	0.28811
1994	15	1.67	1.5516338	0.40063	92.927	0.23994
1995	27	0.59	0.7551708	0.14533	127.815	0.24598
1996	13	0.52	0.7972514	0.22112	153.651	0.42615
1997	28	1.73	4.8859085	0.92335	282.866	0.53457
1998	11	0.06	0.1259172	0.03797	224.378	0.67653
1999	11	0.08	0.1974402	0.05953	240.817	0.72609

Hawksbill Sea Turtles in the Dominican Republic

Yolanda León

Project Description

The data we analyzed came from in-water surveys of juvenile hawksbill turtles in southwestern Dominican Republic. Our estimate of abundance (CPE) was calculated from the number of turtles captured (by hand) and sighted per hour during each survey at a given site. Data were recorded for a maximum of 12 sites. The duration of each survey varied from 0.5 to 3.0 hours. The surveys were conducted during several 5 to 10-day periods in each year from 1996 to 1999. The number of observers employed in each survey varied from 2 to 7.

Analyses

1. CPE's and CV's for our data by year and site are presented in Table 1. We could not compute the CV for every site in all years because in some sites only one survey was conducted. Our CV values were smaller than 100 in most cases, except for 1996. We believe that the data for 1996 were different because 1996 was the first year of our surveys, and personnel training, site selection and team coordination were not comparable to those in subsequent years.

2. Univariate analyses of CPE per year and site: Tests of normality revealed that most samples, except for those in 1996, could be treated as coming from a normal population.

3. Effect of the number of observers per survey: We pooled all survey sites and years to determine if the numbers of observers per survey made a difference in our CPE estimates. CV's declined as the number of observers increased, but all were under 100 except for surveys with 2 observers. The means across different observer numbers did not vary substantially, except again for those surveys with only two observers.

4. Power analyses using the software TRENDS were conducted to determine if we could detect a 10% change in the mean of our population, pooling all sites and using an alpha level of 0.05. Our CV value for the means calculated by year (pooling all sites) was 0.279. The power value was 0.076. Because of the aforementioned problems with the data from 1996, we decided to exclude these data and conduct the power analysis, which lowered our CV to 0.035. Our power value in this case increased to 0.279. However, with just one more year of sampling, assuming constant survey variation, the calculated power for detecting such a trend would increase to 0.832.

Table 1. CPE and CV values for our survey data.

Year	Site	CPE	STD	N	CV	min	max
1996	Bahia	0.2	0.4	11.0	185.6	0.0	1.2
	BeataO	0.0	0.0	8.0	--	0.0	0.0
	Caborojo	1.6	2.1	37.0	134.2	0.0	12.0
	CanalB	0.7	0.9	2.0	141.4	0.0	1.3
	Frailes	0.5	--	1.0	--	0.5	0.5
	LBrigo	0.3	0.5	4.0	200.0	0.0	1.0
	LZo	0.6	0.8	6.0	140.9	0.0	2.0
	MOeste	2.9	3.6	4.0	125.2	0.0	8.0
1997	Bahia	2.0	2.1	5.0	105.2	0.0	5.6
	BeataO	0.0	--	1.0	--	0.0	0.0
	Caborojo	1.7	1.2	22.0	70.3	0.0	4.0
	CanalB	1.0	--	1.0	--	1.0	1.0
	Colita	1.7	0.5	2.0	28.3	1.3	2.0
	Faro	5.0	--	1.0	--	5.0	5.0
	Frailes	2.6	1.0	3.0	38.2	1.5	3.4
	LBrigo	1.1	--	1.0	--	1.1	1.1
	LZo	2.2	1.3	6.0	56.8	0.0	3.7
	MOeste	1.9	0.7	5.0	35.0	0.8	2.6
	Ocrique	1.0	--	1.0	--	1.0	1.0
	PlayaN	3.9	2.8	9.0	73.4	0.0	10.0
1998	Bahia	2.6	0.9	7.0	36.0	1.2	3.6
	Caborojo	0.9	0.9	12.0	97.2	0.0	2.4
	Colita	2.4	0.5	6.0	20.8	2.0	3.0
	Faro	1.6	0.5	4.0	32.5	1.0	2.3
	LZo	1.4	1.2	7.0	88.1	0.0	3.0
	MOeste	3.8	3.3	6.0	86.4	0.0	9.1
	Ocrique	1.0	--	1.0	--	1.0	1.0
	PlayaN	3.1	2.4	9.0	77.0	0.0	7.5
1999	Bahia	2.1	1.1	6.0	50.2	1.0	4.0
	Caborojo	1.8	1.5	6.0	81.1	0.0	4.0
	CanalB	0.0	--	1.0	--	0.0	0.0
	Colita	3.3	3.0	3.0	91.5	0.0	6.0
	Faro	3.0	4.2	2.0	141.4	0.0	6.0
	Frailes	1.9	0.9	2.0	49.7	1.2	2.5
	LBrigo	0.5	0.7	4.0	155.3	0.0	1.5
	LZo	1.1	0.9	4.0	85.1	0.0	2.2
	MOeste	2.2	2.0	3.0	91.7	0.0	4.0
	Ocrique	0.5	--	1.0	--	0.5	0.5
	PlayaN	2.3	2.9	10.0	127.2	0.0	8.0

Juvenile Hawksbills in Buck Island Reef National Monument, St. Croix, US Virgin Islands

Brendalee Phillips

Project Description

In 1994 the National Park Service expanded its Buck Island Reef Sea Turtle Research Program from a nesting study to include an in-water study of the juvenile sea turtle population utilizing the monument as a developmental habitat. To identify resource management needs within the park, the objectives of the study are to identify as many individuals as possible through tagging, determine length of residency, growth rates, habitat utilization, and genetic and gender composition of the population. Snorkel surveys are conducted at random times throughout the year when time and personnel allow. When a turtle is sighted, the location, activity, time, habitat, water depth, and turtle depth are noted and transferred to standardized data sheets upon return to the boat. Free dives are used to attempt hand captures of the turtles when possible. When a turtle is captured, it is taken back to the boat, weighed, measured, tagged, blood samples are taken for testosterone and glucose analysis, tissue samples are collected for genetic analysis, and photographs are taken of the carapace, plastron, head, face, and any identifying characteristics (i.e., wounds, deformities, etc.). To date, 85 turtles have been tagged with NMFS Inconel tags and/or AVID Passive Integrated Transponder tags. There have been 614 recorded hawksbill captures; 465 of those are sightings, and 149 are physical captures.

Analysis

CPE analysis was conducted using SAS software on the six years of Buck Island Reef NM data (1994-1999). The results of that analysis for all sightings and captures per effort hour, sorted by year are as follows:

OBS	YEAR	Mean CPE	Standard Deviation	Standard Error	N	CV	MIN	MAX
1	94	1.14015	0.97400	0.20766	22	85.4269	0.00	4.00
2	95	1.34949	1.13711	0.21489	28	84.2623	0.50	5.00
3	96	1.72222	1.59166	0.31833	25	92.4188	0.25	8.17
4	97	1.19462	0.77643	0.13725	32	64.9937	0.20	4.00
5	98	2.20431	1.88962	0.26723	50	85.7238	0.20	7.00
6	99	1.51984	1.27420	0.19661	42	83.8376	0.50	6.00

The mean number of turtles seen per effort hour across all years was:

N	Mean CPE	Standard Deviation	Minimum	Maximum
6	1.52177	0.39709	1.14015	2.20431

Relative Abundance of Marine Turtles in Mosquito Lagoon, Florida

Jane Provancha

Project Description

The purpose of this ongoing project is to make several assessments of marine turtles in Mosquito Lagoon, Florida. These assessments include condition, sex ratio, species composition, and catch rates. Due to the similarity in sampling methods, it is hoped that current catch per unit effort (CPUE) can be compared with historical CPUE in the same area as well as with other sites in the region. Further evaluation of the data during this special workshop was conducted to determine adequacy of sampling design to detect trends in the population.

Methods

Tangle nets (generally two, totaling 0.461km with depths of about 3.5m) are set at several locations within Mosquito Lagoon for a minimum of 6 hours per site. Water depths range from 1.3 to 2.2m. Nets were originally set in each of four seasons, but after 3 years, the winter season was dropped due to consistently low catch rates. The low catch rates are likely related to temperatures during our winter sampling. Only CPUE data were analyzed for this workshop. Data initially considered for the analyses were for years 1994 through 1999. Basic tendency of the data and simple statistics were derived using SAS. In order to utilize the CVs required in the TRENDS program, the SAS CVs were converted to CVx; Sx/x, Sx = S/n (sqrt). After CVs were established for various groups of data, TRENDS software was applied to determine power of the data as currently collected.

Results of Analyses

Data for each of two species, loggerheads and green turtles, were evaluated by area, by year, and by season. Overall CPUEs were 0.08 and 0.32 for loggerhead and green turtles, respectively. Coefficient of variation (CV) for Mosquito Lagoon green turtles and loggerheads were all relatively high. The CVs were much higher for loggerheads than for green turtles, but CVs for green turtles were also high regardless of year, season, or site. The parameter “site” was dropped from further analysis. The remainder of the analyses were limited to green turtles. The annual mean CPUE for greens were calculated, and the CVs ranged from 185 to 95 as noted in the table below.

The following table shows the annual values associated with greens for “all season” collections.

YEAR	**Mean CPUE**	**Standard Error**	**N**	**CV**	**MIN**	**MAX**
94	0.00	0.00	7	0.0	0.0	0.00
95	0.25	0.12	15	185.5	0.0	1.36
96	0.44	0.19	13	156.6	0.0	1.93
97	0.24	0.09	13	139.0	0.0	1.04
98	0.31	0.08	13	95.0	0.0	0.78
99	0.38	0.12	10	105.6	0.0	1.00

The lowest seasonal CVs for green turtles were from fall and summer seasons as shown in the following table.

SEASON	**Mean CPUE**	**Standard Deviation**	**N**	**CV**	**MIN**	**MAX**
Spring	0.28	0.48	15	173	0.0	1.89
Summer	0.34	0.54	27	158	0.0	1.93
Fall	0.31	0.32	21	104	0.0	1.04
Winter	0.09	0.25	8	282	0.0	0.72

The following table shows the values associated with green turtles for each year using “summer” collections only for 1995-99.

YEAR	**Mean CPE**	**Standard Error**	**N**	**CV**	**MIN**	**MAX**
95	0.54	0.33	5	136.9	0.0	1.36
96	0.85	0.57	3	115.6	0.0	1.93
97	0.37	0.22	3	104.4	0.0	0.78
98	.464	0.18	4	78.62	0.0	0.78
99	0.20	0.20	5	223.6	0.0	1.0

The following table indicates the basic summary statistics for the two time periods tested for 1995-99.

Season	**Mean CPUE**	**Std. Deviation**	**N**	**CV**
Summer	0.48	0.24	20	0.40
All Seasons	0.32	0.08	64	0.29

The new CVs generated for TRENDS included data filtered for green turtles whereby only years 95-99 were used (no turtles were caught in 1994) and by summer vs. all seasons pooled. The program TRENDS was used to explore the power of the analyses of the “summer” data only (CV = 0.40). TRENDS yielded a power of 0.52 with this CV and required a sample size of 15 years when alpha = 0.1. Further testing was completed using the “all seasons” CV (0.29) and choosing exponential, 2 tailed test. The following table indicates results and suggests that a reliable trend (5% annual change) should be detectable within 15 years.

Rate of detectable change	**Alpha**	**Power**	**No. of Years**
0.05	0.1	0.934	15
0.05	0.05	0.867	15

Two additional trials were run for these data along with other projects described in this report, and the results are included in the comparison table at the end of this document. In summary, the data indicate that periods with high CPUE often had high CVs warranting further consideration of refining sampling strategies.

Assessment of Kemp's Ridley Turtles in the Cedar Keys, Florida

Jeffrey R. Schmid

Project Description

The purpose of this project was to characterize the aggregation of Kemp's ridley turtles occurring in the coastal waters east of the Cedar Keys, Florida, using mark-recapture techniques. From 1986 to 1993, the sampling protocol involved fishing three days a week, every other week, during the seasonal occurrence (April-November) of marine turtles in this region. Turtles were captured using 1, occasionally 2, large-mesh tangle nets of 65 m length. Nets were set in the vicinity of the Corrigan Reef oyster bar complex and fished for a 6-hour tide, primarily falling tides.

Analyses

The following analyses were performed with SAS. Number of turtles and netting effort (hours fished by a 1 km net) were pooled by year. Mean annual CPUE (catch per unit effort = turtles/km•hr) and the associated variability were calculated (Table 1).

Table 1. Summary of CPUE estimates and significance of test for normality by year.

Year	Days fished	Mean CPUE (turtles/km•hr)	Standard deviation	Standard error	Normality (p-value)
1986	21	1.95	2.73	0.60	0.0001
1987	27	2.03	2.62	0.50	0.0001
1988	19	4.43	2.60	0.60	0.0616
1989	17	2.96	2.80	0.68	0.0299
1990	18	2.42	2.33	0.55	0.0334
1991	15	1.93	1.45	0.38	0.3698
1992	30	3.10	3.14	0.57	0.0009
1993	19	3.89	3.91	0.90	0.0153
Total		2.84			

Normality of the data was analyzed with Shapiro-Wilk's test for normality and homogeneity of variances was analyzed using Levene's test for equality of variance. Given the non-normality (Table 1) and non-homogeneity of the variances ($F = 2.27$, $p = 0.03$), the Kruskal-Wallis test was used to test for a difference in mean CPUE among years, and there was no significant difference ($\chi^2 = 13.4$, $df = 7$, $p = 0.06$). Linear regression was performed to generate residuals of mean annual CPUE estimates. Mean coefficient of variation of residuals ($CV_{residual}$) was used to assess the precision of the trend and used in the power analyses:

$$CV_{residual} = \frac{MSE}{\text{Total mean CPUE}} = \frac{0.9087}{2.8387} = 0.32$$

CPE for Post-Hatchling Loggerheads Captured at the Western Gulf Stream off Florida

Blair Witherington

Project Description

Trips were made to the western Gulf Stream where habitat of post-hatchling sea turtles occurs. Habitat consists of ‘weed lines’, which are linear accumulations of floating material. When appropriate habitat was found, timed searches were made for turtles from a boat moving at a constant speed of 2.5 kts. Turtles that were observed were captured immediately with a dip net. Each day of sampling consisted of multiple timed sample intervals. In the study, effort was measured in units of time. Two locations (SEB = Sebastian Inlet, and CAN = Cape Canaveral) were sampled in each of two years (1998 and 1999).

Analysis

CPE with associated CVs were calculated by year and by location using a SAS routine. Results of the analysis were used to calculate the number of years needed to detect differences between locations and between years (sample size routine from the SISA Web Site, http://home.clara.net/sisa/samsize.htm). The program TRENDS was used to calculate power and sample size given CVs from the data.

Results

Overall mean CPE = 0.48 turtles/min
Overall CV = 180%
Overall CV of mean = 0.12

CPE data by year and location:

Year	Location	XCPE	STD	STDERR	N	CV	MIN	MAX
98	CAN	0.19633	0.47310	0.06901	47	240.973	0	2.08333
98	SEB	0.33754	0.77377	0.11409	46	229.238	0	3.90909
99	CAN	0.36538	0.57739	0.05408	114	158.025	0	2.00000
99	SEB	1.17470	1.41259	0.20605	47	120.251	0	4.00000

Calculation of sample size from SISA Web Site (two tailed):

CAN vs SEB for 98:

alpha	power			
	0.6	0.7	0.8	0.9
0.1	299	390	516	712
0.05	407	512	655	873

CAN vs SEB for 99:

alpha	power			
	0.6	0.7	0.8	0.9
0.1	26	34	45	62
0.05	36	45	57	76

98 vs 99 at CAN:

alpha	power			
	0.6	0.7	0.8	0.9
0.1	139	182	240	331
0.05	189	238	304	406

98 vs 99 at SEB:

alpha	power			
	0.6	0.7	0.8	0.9
0.1	27	35	47	64
0.05	37	46	59	78

North Carolina Aerial Transect Surveys 1988-1991

Selina Heppell

Project Description

Data collected from aerial transect surveys from 1988 through 1991 off the coast of North Carolina were analyzed using the program DISTANCE. Results are shown in Figure 1 and Table 1.

Figure 1. Coefficient of variation (CV), density (D), and population size (N) expected for all regions combined from an analysis using DENSITY with stratification by year for density and population size estimates, constant detection probability. Read CV and D on left axis, N on right axis.

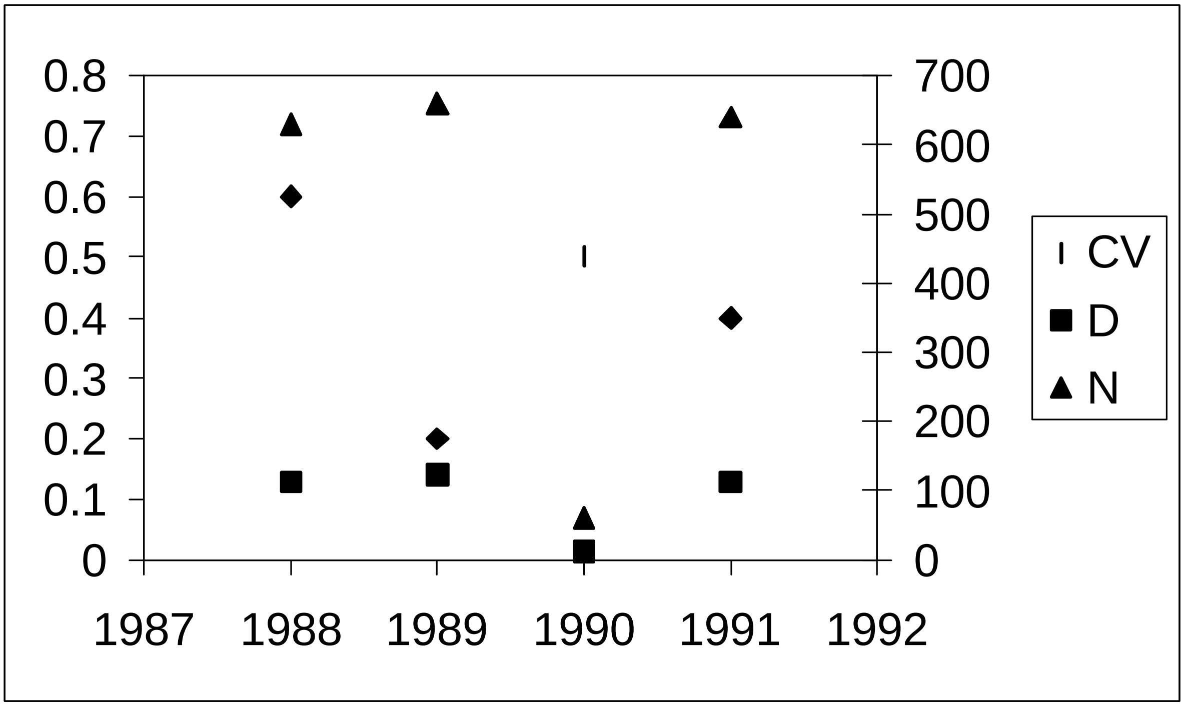

Table 1. DISTANCE results for North Carolina aerial transect surveys 1988-1991. All runs with observation data truncated between 150 and 400 m perpendicular distance, following a model comparison and visual analysis of sighting data. Results for individual years (regions pooled) run as a stratified analysis with constant detection probability.

Data	Sightings/ Obser-vations	Model	AIC	Detection Probability in defined area (p)	Effective Strip Width (ESW)	Density (per km^2)	95% CI (Density)	N	95% CI (N)	CV
All regions all years	140/787	Half-Normal cosine	1489	0.176	70.354	0.089	0.06-0.13	421	286-619	19.87
		Uniform cosine	1488	0.194	77.538	0.081	0.062-0.106	382	292-499	13.7
		Hazard cosine	1487	0.058	23.281	0.320	0.148-0.693	1507	698-3255	40.51
All regions 1988	13/31	Half-Normal cosine		0.176	70.354	0.142	0.048-0.415	665	227-1951	56.93
		Uniform cosine		0.194	77.538	0.128	0.045-0.367	604	211-1725	55.08
		Hazard cosine		0.058	23.281	0.508	0.150-1.713	2385	707-8050	66.99
All regions 1989	100/307	Half-Normal cosine		0.176	70.354	0.147	0.098-0.220	689	459-1033	20.83
		Uniform cosine		0.194	77.538	0.133	0.099-0.178	625	466-839	15.05
		Hazard cosine		0.058	23.281	0.525	0.241-1.143	2469	1134-5375	40.99
All regions 1990	9/321	Half-Normal cosine		0.176	70.354	0.014	0.006-0.032	64	27-152	46.60
		Uniform cosine		0.194	77.538	0.012	0.053-0.028	58	25-133	44.32
		Hazard cosine		0.058	23.281	0.048	0.017-0.141	228	79-663	58.47
All regions 1991	18/128	Half-Normal cosine		0.176	70.354	0.141	0.074-0.271	666	349-1273	33.71
		Uniform cosine		0.194	77.538	0.129	0.071-0.232	605	335-1090	30.48
		Hazard cosine		0.058	23.281	0.508	0.204-1.263	2389	961-5938	48.81
Core Sound all years	94/632	Half-Normal cosine	987	0.143	57.160	0.190	0.121-0.297	47	30-74	22.94
Core Sound 1989	59/225	Uniform cosine	625	0.187	74.958	0.259	0.189–0.357	64	47-88	16.30
North Pamlico all years	19/107	Half-Normal cosine	205	0.196	78.571	0.033	0.010-0.105	54	20-205	62.15
South Pamlico all years	27/48	Half-Normal cosine	296	0.311	124.230	0.035	0.016-0.078	88	40-194	40.72

Estimates of Sea Turtle Abundance from Aerial Surveys

Keith Mullin

Project Description

All sea turtle data were collected during line transect surveys of whales and dolphins. Surveys were conducted from a Twin Otter with bubble windows from an altitude of 750 feet (229m).

Project 1

Study area: 25,659 km^2
Atlantic waters from the mainland shore to the 25 m depth contour from Cape Hatteras, NC to Long Island, NY.

Study period: July and August 1995

Effort: 64 transects (approximately perpendicular to depth gradient) from a random start uniformly covered the area. The set of 64 transects was surveyed 3 times in succession. Total effort was 7,190 transect km.

Abundance:

Species	**n**	**D**	**N**	**CV(N)**
Loggerhead	459	13.97	3,584	0.12
Unid. Chelonid	219	5.82	1,492	0.11
Leatherback	56	1.14	364	0.17
Kemp’s ridley	12	0.32	83	0.28

Where n = number of turtle clusters (clusters averaged 1.1 – 1.5 turtles)
D = turtles/100 km^2
N = number of turtles

Project 2

Study area: 89,856 km^2
Atlantic waters from the mainland shore to inside the western wall of the Gulf Stream from Cape Hatteras, NC to central Florida.

Study period: January and February 1995

Effort: Sixty transects (approximately perpendicular to depth gradient) from a random start uniformly covered the area. These 60 transects were surveyed once. Total effort was 4,356 transect km.

Abundance:

Species	**n**	**D**	**N**	**CV(N)**
Loggerhead	141	13.37	12,013	0.25
Unid. Chelonid	46	3.20	2,874	0.20
Leatherback	9	0.96	862	0.42
Kemp’s ridley	6	0.38	345	0.47
Green turtle	3	0.19	172	0.47

Where n = number of turtle clusters (clusters averaged 1.1 – 1.3 turtles)
D = turtles/100 km^2
N = number of turtles

Project 3 (GulfCet I, Hansen et al. 1996)

Study area: 85,810 km^2
Continental slope and outer shelf waters (100-2000 m deep) of the north-central and northwestern Gulf of Mexico.

Study period: Summer 1992 and 1993, fall 1992 and 1993, winter 1993 and 1994, spring 1993 and 1994.

Effort: 74 transects (approximately perpendicular to depth gradient) from a random start uniformly covered the area. Seventy-four transects were surveyed once each season. Total effort was 49,960 transect km.

Overall Abundance:

Species	**n**	**D**	**N**	**CV(N)**
Loggerhead	12	0.05	41	0.29
Unid. Chelonid	12	0.05	41	0.33
Leatherback	45	0.18	153	0.19
Summer	13	0.20	171	0.43
Fall	11	0.19	159	0.33
Winter	11	0.17	148	0.32
Spring	10	0.16	135	0.28
Kemp’s ridley	2	0.01	7	0.71

Where n = number of turtle clusters (clusters averaged 1.1 – 1.3 turtles)
D = turtles/100 km^2
N = number of turtles

Project 4 (GulfCet II, Mullin and Hoggard 2000)

Study areas: 12,326 km^2 in the Gulf of Mexico shelf waters $<$ 100 m deep and $>$ 18.5 km offshore of the western Florida Panhandle, and 70,470 km^2 of the Continental slope and outer shelf waters (100-2,000 m deep) of the northeastern Gulf of Mexico.

Study period: summer 1996, winter 1997, summer 1997, winter 1998

Effort: 16 and 42 transects (approximately perpendicular to depth gradient) from a random start uniformly covered the shelf and slope areas, respectively. Transects were surveyed once each season. Total effort was 3,652 transect km for shelf waters and 20,880 transect km for slope waters.

Overall Abundance - Shelf:

Species	**n**	**D**	**N**	**CV(N)**
Loggerhead	84	4.08	503	0.20
Summer	39	3.89	480	0.30
Winter	45	4.25	524	0.23
Unid. Chelonid	7	0.34	42	0.45
Leatherback	4	0.19	24	0.48
Kemp’s ridley	4	0.01	12	1.14

Overall Abundance - Slope:

Species	**n**	**D**	**N**	**CV(N)**
Loggerhead	21	0.20	141	0.27
Summer	2	0.03	24	0.77
Winter	19	0.41	286	0.27
Unid. Chelonid	5	0.05	34	0.43
Leatherback	25	0.24	168	0.23
Summer	19	0.33	230	0.58
Winter	6	0.13	90	0.48

Where n = number of turtle clusters (clusters averaged 1.1 – 1.3 turtles)
D = turtles/100 km^2
N = number of turtles

Project 5

Study areas: northern Gulf of Mexico inshore waters (bay, sounds and estuaries), coastal (< 18.5 m deep) and offshore waters (> 18.5 m and < 183 m).
Inshore 26,704 km^2
Coastal 86,439 km^2
Offshore 245,750 km^2

Study periods: western Gulf (1992), central Gulf (1993), eastern Gulf (1994)

Effort: about 22,000 transect km

Overall number of sightings:
Overall abundances have not been estimated. Most of the sightings were in the eastern Gulf of Mexico.

Species	**n**
Loggerhead	234
Kemp's ridley	29
Green turtle	22
Hawksbill	141
Unid. Chelonid	93
Leatherback	30

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Abundance and Distribution of Sea Turtles Estimated from Data Collected during Cetacean Surveys

Debi Palka

Project Description

During 09 July to 17 September 1995, an airplane and two ships surveyed 28,021 km of track lines from Virginia to the lower part of the Gulf of St. Lawrence (563,463 km^2). During 06 July to 08 September 1998, an airplane and ship surveyed 15,886 km of track lines from Delaware to the mouth of the Gulf of St. Lawrence (468,139 km^2). The primary purpose of these surveys was to use line transect methodology to estimate abundance of cetaceans, a secondary purpose was to estimate the abundance of turtles and seals.

Abundance and CV Analyses

Leatherback and loggerhead turtles were detected in both 1995 and 1998. Kemp's ridley turtles were detected in 1998 only. Nearly all of the turtles were detected in waters shallower than 100m, the region surveyed by the NOAA Twin Otter. A few turtles were detected in waters deeper than 100m during the shipboard surveys. Only the aerial leatherback and loggerhead sightings are presented here.

The airplane flew at an altitude of 600 feet (183 m). Four scientists were in the observation team: one was recording the sighting and effort data on a laptop computer, two observers were viewing from bubble windows, one on each side of the plane, and one observer was viewing from a belly window. All cetaceans, turtles, and seals detected were recorded. The angle of declination when a group passed the plane was measured using an inclinameter. This region was stratified into three geographic strata, mid-Atlantic, Gulf of Maine, and Scotian shelf. Nearly all of the turtles were detected in the mid-Atlantic strata. A few were in the Gulf of Maine or Scotian shelf strata. The computer package Distance (Buckland et al. 1993) was used to estimate the abundance of turtles, where abundance was estimated by stratum and species.

For the 1995 data, because of small sample sizes, the detection function was estimated from perpendicular distances from both turtle species within all strata, and the encounter rate and group size was stratum/species-specific. The CV of the abundance was estimated empirically. The uniform model was chosen as the best fit, according to the AIC and the Chi-square test ($p = 0.98$), where the perpendicular distances were left truncated at 0.04 km. The resulting effective strip width (ESW) was 0.22 km. All groups were of size one. From the 11 observed leatherback turtles, the estimated abundance was 1052 (%CV = 38) (Table 1). From the 21 loggerhead turtle sightings, the estimated abundance was 1686 (%CV = 27) (Table 1).

For the 1998 data, because of the larger sample sizes, detection functions were estimated separately for each species over all strata, and the encounter rate and group size was stratum/species-specific. From the 30 leatherback turtle groups, the best detection function model was the uniform model (Chi-square p-value = 0.60), when the perpendicular distance was left truncated at 0.3km and right truncated at 1k. The estimated ESW was 0.28 km, and the mean group size was 1.2 (%CV = 12). The resulting estimated abundance of leatherback turtles was 1174 (%CV = 52) (Table 1). From the 166 groups of loggerhead turtles, the hazard rate model with no adjustments was the best fitting detection function (Chi-square p-value = 0.05), where the perpendicular distances were right truncated at 2000m. The estimated ESW was 0.28 km, and the mean group size was 1.1 (%CV = 12). The resulting estimated abundance for loggerhead turtles was 6007 (%CV = 25) (Table 1).

These abundance estimates are negatively biased due to the assumption that all turtles on the track line were detected (i.e., g(0) = 1). This assumption is invalid due to two factors. One, some turtles are below the surface and so not available to be detected and two, some turtles that were on the surface near the track line were not detected by the observers because turtles are small targets. The probability of detecting harbor porpoises on the track line from the same airplane as estimated from the observers during the 1995 survey was 0.24. If this estimate is approximately the same for turtles, then a less bias abundance estimate would be the product of the estimates in Table 1 and the inverse of 0.24, that is 4.27. Even this estimate of g(0) for turtles is probably negatively biased because turtles spend more time below the surface and turtles are smaller and so more difficult to see than a harbor porpoise.

Table 1. Abundance estimates and %CV of leatherback and loggerhead turtles based on 1995 and 1998 aerial surveys conducted from Virginia to the Gulf of St. Lawrence.

Species	**Year**	**Abundance**	**%CV**
Leatherback	1995	1052	38
	1998	1174	52
Loggerhead	1995	1686	27
	1998	6007	25

Sea Turtle Populations in Florida Bay

Barbara Schroeder and Blair Witherington

Project Description

Population studies of marine turtles in Florida Bay have been ongoing since 1990. Directed capture, using a modified turtle jumping technique has been consistently employed since 1992. Capture methodologies using set nets and drifting nets have been used; however, environmental conditions preclude the use of this gear under most circumstances and hand capture has been selected as the capture method of choice. Line transect surveys were initiated in 1998 with the goal of measuring trends in population abundance over time.

Methodology

Boat-based line transect surveys have been conducted aperiodically beginning in 1998 as part of a larger population study of marine turtles in Florida Bay. Transect surveys are conducted in two basins in the western central portion of Florida Bay. Ten and nine transects, respectively, are surveyed in portions of Rabbit Key and Arsnicker Basins. Transects are sampled without replication within a particular sampling period. Survey area of Rabbit Key Basin is 3.43 km^2 and Arsnicker Basin is 2.24 km^2. Two observers, stationed on the stern platform (eye level approximately 15 feet above water line) sight either side of the track line. Perpendicular distances of observed turtles are calculated using the sighting angle as measured by the observer with an inclinometer. Transect surveys are conducted only when environmental variables are optimal (e.g., sun angle, % cloud cover, sea state). Data are collected on three species, loggerhead, green turtle, and Kemp's ridley. Three dedicated line transect surveys have been conducted to date in each survey area (basin).

Analyses

Density and Abundance Analyses

DISTANCE software was used to determine the detection function and to estimate density and abundance. The small sample sizes required some pooling of data. The detection function was first examined by combining data by species over all surveys. Four models were evaluated: half-normal/cosine, uniform/ cosine, hazard-rate/cosine, and hazard-rate/hermite polynomial. The results of the models selected as the best fit by species are:

Species	**Model**	**AIC**	**%CV**	**p**	**ESW (m)**
Loggerheads	Hazard/hermite	190.02	18.46	0.50984	15.295
Green turtles	Half-normal/cosine	128.66	16.48	0.50368	15.110

The effective strip widths and detection probabilities are quite similar, and, therefore, subsequent analyses combined species observations to estimate the detection function with post-stratification by species. Sample sizes were too small to further stratify by survey basin or by sample year. Density and abundance estimates were calculated only for loggerhead and green turtles; samples sizes for Kemp's ridley were too small to provide meaningful results. Of the four models evaluated (listed above), the default model (half-normal) resulted in the best fit. The following results were obtained:

	Density Estimate (per km^2)	Abundance	%CV	df	95% Confidence Interval (Density)
Loggerheads	11.37	64	21.48	85	7.45 - 17.34
Green turtles	7.58	43	33.34	67	3.96 - 14.49

NOTE: While the analyses provide information relative to the selected survey area, it cannot be assumed that these results would be applicable to Florida Bay as a whole. Since the sampling areas are relatively small, they may say little about the population on a broader scale, but may have value when addressing issues related to the more limited scale.

Green Turtles at Conception Island Creek, Bahamas

Karen Bjorndal and Alan Bolten

Project Description

This study is based on capture-recapture. Immature green turtles are surveyed annually in Conception Island Creek, Bahamas. Turtles are captured by jumping on them from an outboard motor boat. Turtles are tagged, measured and released at site of capture. Data from 1989 through 1998 are analyzed here. The surveys are ongoing in collaboration with Steve Connett and his students at St. George's School, Newport, Rhode Island.

Analysis

We did not divide the turtles into attribute groups.
Using the software MARK, we tested models with

1. Time-dependent survivorship, time-dependent capture probability
2. Time-dependent survivorship, constant capture probability
3. Constant survivorship, time-dependent capture probability
4. Constant survivorship, constant capture probability

The model with best fit was time-dependent survivorship with constant capture probability.

Using the software RELEASE (available within MARK), we tested the goodness of fit (GoF) of the full model. The model passed Test 2 and Test 3:

Chi-square = 24.9315, df = 22, P = 0.3004.

We then assessed the GoF of our best model (time-dependent survivorship, constant capture probability) with a bootstrap test in MARK with 1000 iterations. The model had a good fit; p = 0.771.

Results are shown in Table 1 and Figure 1. In Figure 1, it appears that population abundance has varied, with no clear increase or decrease.

Because the values are linked between years, and thus are auto-correlated, it is not appropriate to evaluate the trend with least squares linear regression, as in the TRENDS software. However, to compare capture-recapture with the other methods, we included our results in the workshop summary table of the results of Trial A and Trial B.

Table 1. Number of green turtles captured each year (ni = Ri) and estimates of abundance (Ni) for green turtles in Conception Creek, Bahamas.

Year	ni = Ri	Ni	Ni(sd)	LCL	UCL	CV
1989	22					
1990	20	37	2.97	31	42	0.081
1991	28	51	4.16	43	59	0.081
1992	32	59	4.75	49	68	0.081
1993	21	38	3.12	32	45	0.081
1994	27	49	4.01	42	57	0.081
1995	42	77	6.24	65	89	0.081
1996	25	46	3.71	39	53	0.081
1997	22	40	3.27	34	47	0.081
1998	31	57	4.60	48	66	0.081

Figure 1. Estimates of abundance of green turtles in Conception Creek (solid line) and 95% confidence intervals (dashed lines).

Summary Tables and

Conclusions and Recommendations

Comparison of Methods to Estimate Trends in Population Abundance

The following summary table was developed from discussions in which the different methods of estimating trends in population abundance--CPE, transect, and capture-recapture--were compared.

Three questions were developed as a basis for the comparison (see last column):

1. Does the method measure a trend?
2. Does the method give an estimate of abundance?
3. Does the method provide access to turtles so that other demographic information can be obtained?

Also cost of the method – expressed in terms of cost per turtle, cost per hectare, or total annual cost of study – was identified as a factor that should be quantified for future comparisons.

	Variability							
	within year	among years	Cost	Labor	Frequency of sampling	Area covered (average study)	Specificity (species, size)	Questions best answered
CPUE								
tangle net	med	high	low	med	annual for Q3 less for Q1*	small	all	1,3
trap net	low	med	med	med	annual for Q3 less for Q1	med	all	1,3
individual capture	low	high	low	med	annual for Q3 less for Q1	small	all	1,3
trawl	high	high	high	low	annual for Q3 less for Q1	med	all	1,3
Transect								
aerial	high	low	high	low	periodic	large	species, size	1,2
ship	high	high	high	med	periodic	med	species, size	1,2
Capture-recapture								
net	med	med	med	high	annual	small	all	1,2,3
individual capture	low	low	low	med	annual	small	all	1,2,3
trawl	high	high	high	med	annual	med	all	1,2,3

* Q1 = Question 1, Q3 = Question 3

Comparison of Ability to Detect Trends

Two trials, using the TRENDS program, were run with the data from each participant with the following parameter specifications:

TRIAL A: Solve for Minimum Detectable Annual Rate of Change
Parameters:

- Each project's duration
- Each project's coefficient of variation (CV)
- Type of growth = exponential
- Sign = negative; 2-tails
- Alpha = 0.2
- Power = 0.9

TRIAL B: Solve for Minimum Duration (years) Required to Detect Annual Change of 25%
Parameters:

- Each project's coefficient of variation (CV)
- Annual change = 0.25
- Type of growth = exponential
- Sign = negative; 2-tails
- Alpha = 0.2
- Power = 0.9

The results of these trials are presented in Table 1. These standardized trials allowed comparison among projects and methods. However, these results should be considered preliminary—calculations, particularly of CVs, were, of necessity, done rapidly during the workshop.

Figure 1 illustrates the general relationship between the detectable rate of annual change and the CV among years (or process error) for studies with a duration of either 5 or 10 years.

Table 1. Summary of project descriptions and results of Trials A and B described above.

	Project Description						Results	
Analyst	Gear	Location	Years	Species	Duration	CV	Trial A	Trial B
CPE								
Balazs	bullpen	Hawaii	1982-99	Cm	18	0.68	0.06	11
Diez	sight/capture	Mona Island Puerto Rico	1992-99	Ei	8	0.28	0.11	7
Ehrhart & Redfoot	tangle net	Indian River Lagoon, FL	1989-98	Cm	10	0.11	0.03	4
	tangle net	Indian River Lagoon, FL	1989-98	Cc	10	0.08	0.02	4
	tangle net	Offshore reef, FL	1989-99	Cm	11	0.13	0.03	4
Epperly	poundnet	North Carolina	1995-97	Lk	3	25	--	--
	poundnet	North Carolina	1995-97	Cm	3	5.45	--	--
	poundnet	North Carolina	1995-97	Cc	3	3.32	--	--
Henwood	trawl	Gulf/Atlantic	1978-81	Lk	4	41.8	--	--
	trawl	Gulf/Atlantic	1978-81	Cc	4	6.43	--	--
Landry	tangle net	Sabine Pass, TX	1992-99	Lk	8	0.39	0.14	8
Leon	sight/capture	Dominican Republic	1997-99	Ei	3	0.04	0.13	3
Phillips	sight/capture	Buck Island USVI	1994-99	Ei	6	0.05	0.03	3
Provancha	tangle net	Mosquito Lagoon Florida	1995-99 Summers	Cm	5	0.40	0.35	8
	tangle net	Mosquito Lagoon Florida	1995-99 All seasons	Cm	5	0.29	0.26	7
Schmid	tangle net	Cedar Keys Florida	1986-93	Lk	8	0.32	0.12	7
Witherington	sight/capture	Florida Atlantic coast	1998-99	Cc	2	0.12	--	4
Line Transects								
Heppell	aerial	North Carolina	1988-91	Spp	4	0.25	0.37	6
Mullin	aerial	Cape Hatteras, NC to Long Island, NY	1995	Cc	1	0.12	--	4
	aerial	Cape Hatteras, NC to Long Island, NY	1995	Spp	1	0.11	--	4
	aerial	Cape Hatteras, NC to Long Island, NY	1995	Dc	1	0.17	--	5
	aerial	Cape Hatteras, NC to Long Island, NY	1995	Lk	1	0.28	--	7
Palka	aerial	Mid-Atlantic US	1995,98	Dc	2	0.38	--	8
	aerial	Mid-Atlantic US	1995,98	Cc	2	0.26	0.65	6
Schroeder	boat	Florida Bay	1998-99	Cc	2	0.21	--	5
	boat	Florida Bay	1998-99	Cm	2	0.33	--	7
Capture - Recapture								
Bjorndal & Bolten	individual capture	Conception Island, Bahamas	1989-98	Cm	10	0.08	0.02	4

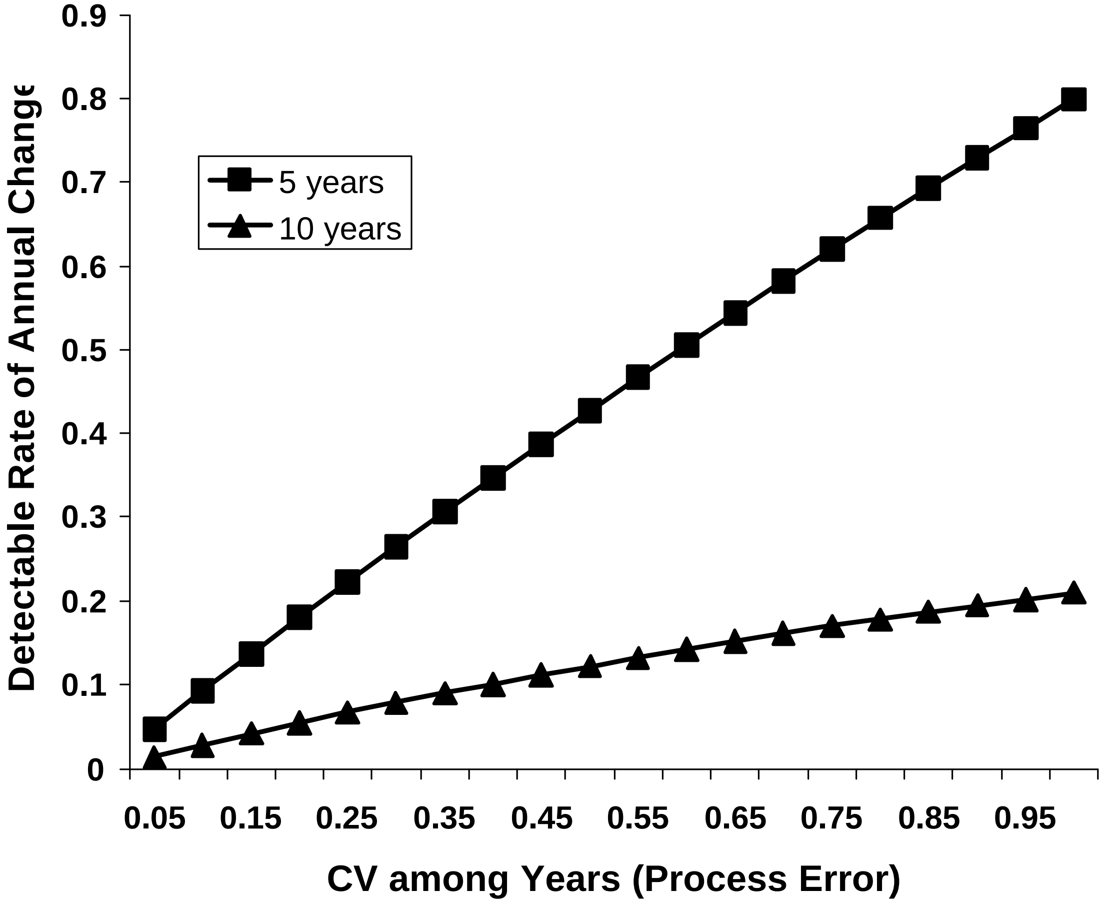

Figure 1. Relationship, determined with the program TRENDS, between detectable rate of change and CV among years for studies with duration of 5 or 10 years, with alpha = 0.2 and power = 0.90.

Conclusions and Recommendations

Although the duration required to detect trends in relative abundance varied, all techniques reviewed during the workshop appeared to be feasible for evaluating trends in relative abundance, with the exception of trawl surveys. However, the size of the area to which the trends can be applied directly is small in most studies. Even within small areas such as a particular lagoon or estuary, sites have not been chosen in a way that allows a trend to be inferred within that system. This was identified as the major shortcoming of most data at the workshop for monitoring purposes, but a shortcoming that could be overcome by the addition of new sites.

For most applications, trends in relative abundance are sufficient; trends in absolute abundance are not required. Estimates of absolute abundance, however, are needed to evaluate activities that may incidentally or intentionally take turtles. Information on population dynamics (e.g., recruitment, growth, survivorship) is critical for reliable interpretation of trends and evaluation of incidental or directed take.

Assessment of trends must be a conservative process. The potential for error by failing to detect a trend (Type 2 error) must be emphasized in addition to the potential for error by concluding a trend exists when it does not (Type 1 error).

A series of study sites that represent all lifestages and the range of the population (i.e., not limited to U.S. waters) is needed to monitor stock-specific trends.

Current sites should be classified as **C**omprehensive, **A**dequate, and/or **R**epresentative (CAR analysis) to identify what additional sampling sites are needed. The analysis should also address the question: Is it better to have fewer studies with thorough sampling (lower CV and/or demographic parameters) rather than more, less-thorough studies?

Methods do not have to be standardized among studies, if internally consistent techniques have been employed in each study and if trends are calculated for each study. The trends can then be compared and/or combined among studies—perhaps using meta-analysis. The potential for using the trend from each study as a replicate in analyses should be evaluated.

Methods for analyzing trend data (with emphasis on CPE data) need to be explored and improved (e.g., nonparametric bootstrap methods).

Studies are needed to determine the relationship between CPE and estimates of density. CPE does not always increase with population density, and if CPE does increase with population density, the relationship is not always linear.

Studies are needed to determine the relationship between capture effort (e.g., net length, soak time) and rate of capture. For example, the relationship between length of a tangle net and number of turtles captured is not necessarily linear.

Studies are needed to estimate G(0) – the fraction detected on track line – for line transects.

The cost/benefit ratios of the different monitoring techniques are of primary concern and should be evaluated.

To the extent possible, a sampling strategy should be employed in which intensive sampling is focussed on turtle-dense areas and less intense sampling is conducted in less turtle-dense areas. To this sampling regime, very large scale surveys—aerial or trawling—could be added if appropriate for the habitat and/or species. This sampling strategy would help to ensure that changes in populations are recognized. For example, if the population decreases, the abundance of turtles in the optimal turtle-dense areas may remain constant while abundance of turtles in surrounding, sub-optimal areas decline. If only the turtle-dense areas are being monitored, the population decline may not be detected.

Studies designed to evaluate trends and/or abundance of sea turtles on foraging grounds, should:

1. Emphasize relative trends and evaluate probability of trend detection based on CV and duration of study.
2. Use standardized techniques throughout the study. If changes are essential, effects of changes on CPE or other measures of abundance should be evaluated.
3. Identify biases within the study (e.g., turbidity in visual sampling, population sinks) and determine if biases change over time.
4. Evaluate the size of the area for which the study site is representative.
5. Determine genetic stocks of the turtles.
6. When feasible, evaluate absolute abundance and other population dynamics parameters (e.g., recruitment, growth, survivorship).
7. When feasible, employ a sampling strategy in which areas with higher turtle densities are sampled intensively and suboptimal habitats with lower turtle densities are sampled less intensively to improve probability of detecting changes in abundance. Determine the appropriate division of sampling effort between areas with high turtle density and those with low turtle density.