

RESEARCH
PAPER



Are green turtles globally endangered?

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ABSTRACT

Aim To examine the exploitation, recovery and current status of green turtles (*Chelonia mydas*) nesting at Ascension Island.

Location Ascension Island (UK) (7°57' S, 14°22' W), South Atlantic Ocean.

Methods We analysed records of the harvest of green turtles nesting at Ascension Island between 1822 and 1935, illustrating the decline in numbers over this period. Using a deterministic age-class structured model we predict the initial number of breeding females present in the population prior to the recorded harvest and compare this to our estimate of the current population based upon our recent annual surveys (1999–2004).

Results Prior to 1822 we estimate the nesting population of green turtles to have been at least 19,000–22,000 individuals in order for the population to have survived the level of harvest recorded. From recent data (1999–2004), we estimate the current breeding population of green turtles at this site to be 11,000–15,000 females. Our results illustrate a dramatic recovery of the population, which is still increasing exponentially and shows no evidence of slowing, suggesting it has not reached 50% of its carrying capacity.

Main conclusions We estimate that, since the 1970s, the Ascension Island population of green turtles has increased by 285% and question the recent listing of this species as endangered by the IUCN (World Conservation Union), in particular in the Atlantic Ocean, where 75% of the populations assessed by the IUCN are increasing. Indeed, we estimate the global population of this species to be in excess of 2.2 million individuals. We suggest that the IUCN's global listing process detracts attention from those populations that are truly threatened with extinction and should not, in its present form, be applied to globally distributed long-lived species such as marine turtles.

Keywords

Ascension Island, *Chelonia mydas*, green turtle, IUCN, Red List.

INTRODUCTION

The IUCN (World Conservation Union) designation of species within their *Red List of Threatened Species* is one of the central tenets upon which strategies to preserve global biodiversity are based. When the IUCN listed the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered (declined by > 80% over the last three generations; www.redlist.org) in 1996, it led to a heated debate regarding the lack of transparency and supporting docu-

mentation of the listing (Brackett, 1997; Lapointe, 1997; Mrosovsky, 1997, 1998; Meylan, 1998). In 2004, the IUCN's Species Survival Commission (SSC) accepted the IUCN–Marine Turtle Specialist Group's (MTSG) latest Red List assessment for the green turtle (Seminoff, 2004a). The MTSG collated, and made available, data from 32 index nesting sites of the green turtle and estimated that over the last three generations (c. 130 years), female green turtles have declined by 48–67% (Seminoff, 2004a), fulfilling the IUCN criteria for listing as endangered (having globally declined by

> 50% over the last three generations), and being 'at very high risk of extinction in the wild'. Although the criteria for classification of species within the *Red List* have changed since the previous assessment of the green turtle in 1996, this species has been listed as endangered since its first assessment by the IUCN in 1982.

In some regions of the world, local extinction of green turtles is likely to occur as a result of factors such as by-catch, directed harvest and habitat loss. For example, at Aves Island (Venezuela), one of the eight Atlantic populations used in the IUCN assessment, nesting females are estimated to have declined by more than 90% since 1879 (Seminoff, 2004a). In contrast, many green turtle nesting populations are actually on the increase as a result of direct conservation action and are not under threat of extinction. One such population is that at Ascension Island, a remote oceanic island in the South Atlantic Ocean (UK 7°57' S 14°22' W). Here, female green turtles were harvested for meat since the island's discovery in the 16th century. The island remained uninhabited until the British took possession and was garrisoned by the Royal Marines in 1815. The harvest of females was documented until the 1940s when it had become commercially unviable and ceased (Huxley, 1999).

METHODS

We have collated archival harvest records (Letters to the Admiralty: Public Records Office; Parsons, 1962; Huxley, 1999) which illustrate the decline in the harvest of this species (Fig. 1). Although a substantial proportion of records (1864–1908) are missing, correspondence indicates that the harvest continued throughout and that reduced levels of harvest were a result of a lack of females.

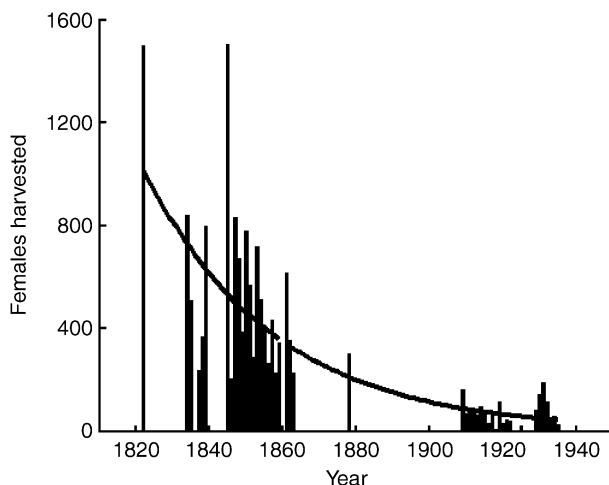


Figure 1 The number of females harvested at Ascension Island, 1822–1935 for years when data were available. See Methods for description of the model. Data are from archival records and Letters to the Admiralty from the Commander of HMS Ascension (as it was then known) and the Eastern Telegraph Company (ETC) when it took administrative control of the island in 1922. In 1935 the ETC did not renew their concession to harvest turtles, it being considered uneconomical. After this date small numbers of turtles were harvested for local consumption only.

Using a deterministic age-class structured model, we predict the initial number of breeding females present in the population prior to 1822 (see description of the model below).

A common method to assess sea turtle populations is to infer the number of nesting females from an estimate of the number of clutches laid. In a given season, females normally lay multiple clutches. In their calculations the MTSG (Seminoff, 2004a) estimate that each female lays an average of three clutches, which we followed in our analysis for direct comparison. In addition, females do not nest every year, but at Ascension Island typically return to breed after an interval of 3–4 years (Mortimer & Carr, 1987) and hence the estimates given are for the annual nesting population. As a large proportion of crawls do not result in successful laying, nesting activities are assessed from track and pit morphology. For three complete seasons (1999–2001) we monitored activities on all nesting beaches at Ascension Island and assessed a large proportion of activities to determine adult emergence success (number of activities resulting in deposited clutches) for each beach (Godley *et al.*, 2001). Using the mean emergence success and mean proportion of activities that occurred on each nesting beach during these years we were able to estimate the total number of clutches laid in subsequent years (2002–04) and from previous studies from track counts on major beaches alone (1977 and 1978: Mortimer & Carr, 1987; unpublished data, Ascension Island Administrator's Office, 1980, 1981 and 1990).

Description of the model

We modified a deterministic age-class-structured, birth-pulse and post-breeding census matrix projection model developed previously to describe the Great Barrier Reef population of green turtles (we have used MODEL1 as a template; Chaloupka, 2001) to estimate the minimum population size of green turtles at Ascension Island that must have been present in 1822 to allow for the level of harvesting that occurred. We have modified the transition matrix (TM) in the model (Chaloupka, 2001) to be representative of the system at Ascension Island and in order to consider only females (as only females were harvested). We used the same growth and survival probabilities as in the Chaloupka (2001) model but modified some fecundity values, as these are likely to differ from the situation in the Great Barrier Reef system for which the model was designed originally. We calculated the fecundities in an identical manner but used the following values specific to Ascension Island: PSR (probability of being female) = 0.75 (derived from previous papers; Broderick *et al.*, 2001a; Godley *et al.*, 2002); EPC (mean eggs per clutch) = 127.5 (Hays *et al.*, 1993); CPS (mean number of clutches per season) = 3; RMI (mean re-migration interval): we ran two versions of the model, with RMI = 3 and 4; %MATURE_{*i*} (age-class-specific proportion of sexually mature individuals: we used values of 0.5 and 1.0 for S5 and S6, respectively, taken from Table 1 in Chaloupka, 2001). We then calculated the associated fecundity (f_i) and fertility values (F_i) according to the following formula taken from Chaloupka (2001):

$$f_i = \text{PSR} \times \text{EPC} \times \text{CPS} \times \text{RMI}^{-1} \times \% \text{MATURE}_i$$

$$F_i = (f_i \times P_i) + (f_i \times G_i)$$

Table 1 Transition matrix (TM) structure and values used in our simulations

	Age-class	S1	S2	S3	S4	S5	S6
Structure	<i>S1 (egg-neonate)</i>	P1	F2	F3	F4	F5	F6
	<i>S2 (pelagic individual)</i>	G1	P2	0	0	0	0
	<i>S3 (benthic individual)</i>	0	G2	P3	0	0	0
	<i>S4 (sub-adult)</i>	0	0	G3	P4	0	0
	<i>S5 (maturing adult)</i>	0	0	0	G4	P5	0
	<i>S6 (adult)</i>	0	0	0	0	G5	P6
Values	<i>S1</i>	0	0	0	F4	F5	F6
	<i>S2</i>	0.4394	0.5704	0	0	0	0
	<i>S3</i>	0	0.0741	0.8413	0	0	0
	<i>S4</i>	0	0	0.0391	0.8405	0	0
	<i>S5</i>	0	0	0	0.0069	0.7782	0
	<i>S6</i>	0	0	0		0.1700	0.9482

Specific values for F4, F5 and F6 are shown in methods.

where G_i is the probability of surviving and growing to the next age-class and P_i is the probability of surviving and remaining in the i th age-class. Note that $[F4, F5, F6] = [0.3299, 53.4639, 90.6716]$ and $[0.2474, 40.0980, 68.0037]$ for RMI = 3 and 4, respectively. To adapt the model to consider only females we then divided the fertility values by PSR and multiplied G_1 by PSR in order to consider only female eggs that matured to G_2 ; the values in Table 1 are the final and actual values we used in our model. Because the model is entirely deterministic, each total population size has a unique stable age-class structure in terms of the proportion of individuals in each age-class. This allowed us to start our simulations with a population of any size at a stable age-structured equilibrium.

We incorporated harvesting into our model by removing a certain number of sexually mature females ($S4, S5$ and $S6$ from Table 1) after each year's breeding event. Although harvesting of immatures from this population might be pertinent at other sites, we have no estimates of this activity. Should such activity have occurred it would require a larger initial population size in 1822, hence the figures we give are unlikely to be over-estimated. We estimated the level of harvesting across all years, from 1822 until commercial harvest was abandoned in 1935, by fitting a curve using a logistic model to the harvest data set ($F = 126.65$, residual mean square = 0.40, $P < 0.0001$). We chose to use the output of this regression model in terms of females harvested per year as input for the model rather than use a mixture of real and estimated data.

At the end of each breeding year a number of breeding females were removed from the i th age-class (H_i) in $S4, S5$ and $S6$. The number of females removed from each of the three breeding groups was proportional to the number of individuals in that group: given by $H_i = H_Y \times N_i / (N_4 + N_5 + N_6)$, where H_Y is the total number of females harvested in year Y (taken from the fitted logistic curve Fig. 2) and N_i is the number of individuals in the i th age-class.

The population was then modelled over successive years beginning in 1822 for a range of initial population sizes. The out-

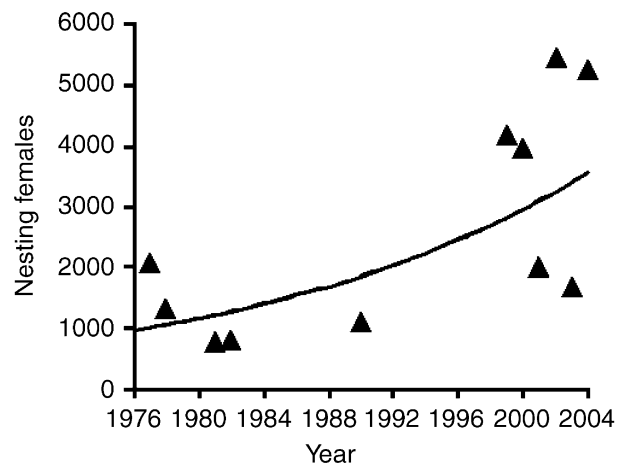


Figure 2 The number of green turtles nesting at Ascension Island, assuming each female lays three clutches per annum. We analysed published data from 1977 and 1978 (Mortimer & Carr, 1987) and unpublished data collected by the Ascension Island Administrator's Office for 1981, 1982 and 1990 in addition to data from our current study (1999–2004).

put of the model is a simple Boolean answer, which either states that the population of turtles was still alive in 1935 when harvesting ended, or whether it was extinct. Clearly, the larger the initial population, the more likely the population would survive the intensive harvest regimen. We started simulations with an 1822 population size of 1000 breeding females and incrementally increased this initial starting value by 1000 individuals until the population survived until 1935.

RESULTS

Our simulations predict that the initial total number of breeding females at Ascension Island in 1822 must have been greater than 19,000 when RMI = 3, and greater than 22,000 when RMI = 4, in order for the population to have survived to 1935. Dividing these

values by their respective RMI values gives us estimates of approximately 5500–6300 individuals for the minimum number of females breeding annually in 1822. It is important to consider this model in light of the parameters used as inputs. The remigration interval (RMI) strongly affects the predictions, but by using values recorded for this population [Mortimer & Carr (1987) recorded 75% of females in their study returning after RMIs of 3 or 4 years with only 7% returning after 2 years, 0% after 1 year and 16% after 5+ years], we are confident that we have not over-estimated the true value. The area under the fitted harvest curve (total number of females harvested) in Fig. 1 is also likely to alter the model predictions. Given that the fitted curve drastically under-estimates two large catches of 1500 females pre-1850, it is likely that we have again made a conservative estimate of the initial population size, unless the catches for the missing years were dramatically smaller. This deterministic model assumes density, and temporally (e.g. varying climatic factors), independent fecundities and survival probabilities (Table 1). Clearly, if any of these values are reduced at large population sizes, such as those modelled as initial starting values for our simulations, then we will again have under-estimated the minimum population size predictions for 1822. Given that turtles worldwide are currently believed to be at levels below their carrying capacities, and have been for some time, it is realistic to assume density independence in our model. Furthermore, increasing values in the transition matrix (Table 1) will reduce our estimate of population size in 1822, and reducing these values will increase our estimate. Given that these figures are largely drawn from present-day data sets (Chaloupka, 2001), when anthropogenic threats are greater than in the past (e.g. industrialized fishing activities and widespread habitat degradation), these parameters probably err on the lower side of the true values for the 1800s and hence our estimate is once again likely to be a lower one.

From the detailed nesting data that we have collected (1999–2004) we estimate that there are presently 3800 (± 1550 , $n = 6$ years) females nesting annually at this site, giving us an estimate of the current total breeding population of 11,000–15,000 females and making this the second largest rookery for this species in the Atlantic Ocean. We have also analysed data collected (1977, 1978, 1980, 1981, 1990) by others and, using our methodology, estimate that during this period an average of 1248 females nested annually. Collectively, these data indicate that the number of nesting individuals may have increased over the past three decades at Ascension Island by as much as 285% (Fig. 2). Growth is described more effectively by an exponential than a linear model. More interestingly, applying logistical and exponential regression models to our data produced identical results ($F_{1,9} = 9.15$, residual mean square = 0.28, $P < 0.014$), suggesting that there is no evidence that the growth rate is slowing through competition effects. Because the point of inflection of the logistic model occurs at 50% of the carrying capacity, this suggests that (despite recent spectacular growth) the population still has the potential to at least double in size. Our estimate of at least 19,000–22,000 breeding females in 1822, more than 150% of the present-day estimate of 11,000–15,000, lends support to this theory.

DISCUSSION

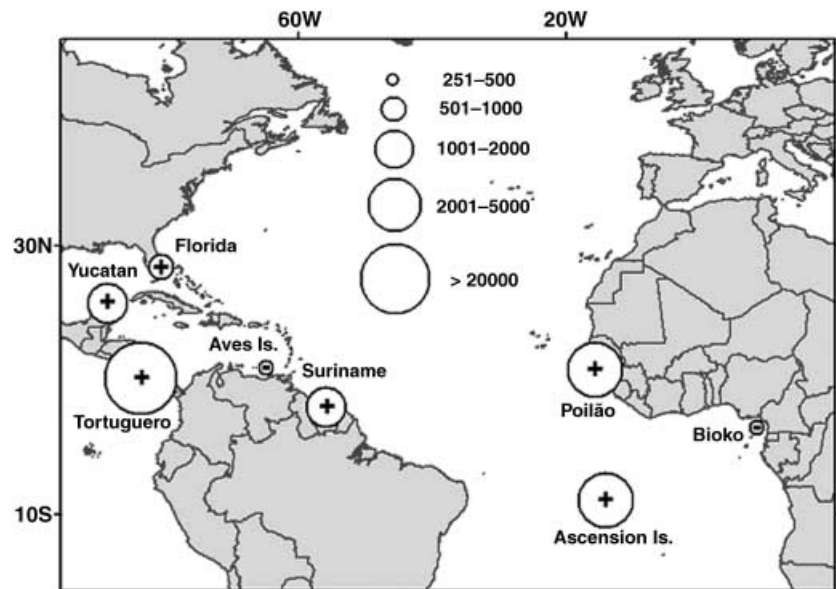
The status of the Ascension Island population at the nesting grounds is now favourable, with adults and eggs both afforded a high level of protection. Unlike the leatherback turtle (*Dermochelys coriacea*), that ranges widely in the open ocean to forage (Ferraroli *et al.*, 2004; Hays *et al.*, 2004), and whose populations have declined drastically in the Pacific (Spotila *et al.*, 2000), the relatively localized inshore foraging grounds of the herbivorous green turtle may enable this species to be protected more effectively in the coastal foraging habitat with the designation of protected areas. Although interactions with coastal fishers may occur, large scale industrial by-catch from pelagic long-lines is less likely to cause a major impact upon this species, unlike the carnivorous leatherback and loggerhead (*Caretta caretta*) turtles (Lewison *et al.*, 2004). Satellite tracking has revealed the main foraging grounds of the Ascension green turtle to be the coastal waters of Brazil (Luschi *et al.*, 1998), where small-scale traditional fisheries for marine turtles once existed and a pioneering conservation programme has dramatically reduced by-catch (Marcovaldi *et al.*, 1998). Notwithstanding, the current Ascension Island population of green turtles appears to be a fraction of that which existed centuries ago and to strive towards pre-exploitation levels may be a realistic goal. Irrespective of this, this population is unlikely to decline and go extinct in the near future under current circumstances.

Data collated by the MTSG from several other Atlantic populations, such as Tortuguero (Costa Rica), Florida (USA) and the Yucatan Peninsula (Mexico), illustrate that the numbers of green turtles nesting have more than doubled over the past 20–30 years (Seminoff, 2004a). In fact, numbers of green turtles are increasing at six of the eight chosen Atlantic index sites (Fig. 3), and over the period analysed have increased by 10–14%. Furthermore, examples of populations increasing are not restricted to the Atlantic, with increases in green turtle populations reported at major sites elsewhere (e.g. Australia: Chaloupka & Limpus, 2001; Hawaii: Balazs & Chaloupka, 2004).

Although the latest MTSG assessment of the green turtle has only recently been accepted, the assessment was made on data prior to 2002. Already this assessment is out of date. For example, while the MTSG assessment calculates the Tortuguero population of green turtles as having increased by 100% since the 1970s, more recent data (Troëng & Rankin, 2005) suggest that an increase of nearer 250% has occurred. Similarly, our estimate of an increase in 285% at Ascension Island is significantly higher than the 39% estimated by the MTSG. Large inter-annual variation recorded in green turtle populations (Broderick *et al.*, 2001b) may account for a proportion of this discrepancy, and highlights the need for continued long-term monitoring for this species. However, this also highlights the rapidity with which turtle populations can recover (Balazs & Chaloupka, 2004; Hays, 2004), the difficulty in collating global data sets and the need for continual regional assessments.

Elsewhere, many green turtle nesting populations are in decline and should be classified as locally endangered, or in some cases critically endangered. Indeed, the green turtle population

Figure 3 The increasing (+) or decreasing (–) status of the eight Atlantic nesting populations of green turtles chosen as index sites by the MTSG. Key indicates the estimated size of the annual nesting population by the MTSG. Locations: Poilão is an offshore island of the Bijagos Archipelago, Guinea Bissau; Bioko is a small island off the coast of Equatorial Guinea. Other locations are described in the text.



in the Mediterranean has been listed by the IUCN as a critically endangered subpopulation (Hilton-Taylor, 2000). So why have some of the subpopulations that have increased not been listed as vulnerable (declined by > 30%) or indeed de-listed? Admittedly, it is difficult to define discrete subpopulations in such a wide-ranging species, but it would seem that a better alternative to listing the species as globally endangered would be regional listing, at least at the level of ocean basin until subpopulations can be defined reliably (Mrosovsky, 2003, 2004; Seminoff, 2004b). There may be a belief among some that declaring a species as globally endangered will afford it a higher level of protection and lead to an increase in conservation effort globally. By listing the green turtle as endangered, the IUCN is suggesting that this species has a 'very high risk of extinction in the wild'. Here we have demonstrated conclusively that this is not the case. Indeed, using the stable age-class distributions derived from our model, and estimates of nesting females from the 32 index sites assessed by the MTSG for their global assessment (Seminoff, 2004a), we estimate the population of green turtles (including all life stages) at these 32 sites alone, to be in excess of 2.2–2.6 million individuals. The green turtle is not at risk of global extinction, but many subpopulations will become extinct if directed action is not taken. The IUCN *Red List of Threatened Species* is a central foundation for efforts to preserve global biodiversity and provide key information for policy makers at all levels; however, the current global listing process detracts attention from those populations that are truly threatened with extinction.

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BIOSKETCHES

Annette Broderick and **Brendan Godley** coordinate the Marine Turtle Research Group (MTRG) <www.seaturtle.org/mtrg/>, a group of scientists and students conducting fundamental and applied research on marine turtles and supporting local capacity building and environmental awareness efforts.

Tara Pelembe and **Robert Frauenstein** were based on Ascension Island and assisted with fieldwork, as did **Fiona Glen**, former MTRG doctoral student.

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