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Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: multiple indices describe population status

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From June 1995 to August 2002 we assessed green turtle (*Chelonia mydas*) population structure and survival, and identified human impacts at Bahía de los Angeles, a large bay that was once the site of the greatest sea turtle harvest rates in the Gulf of California, Mexico. Turtles were captured live with entanglement nets and mortality was quantified through stranding surveys and flipper tag recoveries. A total of 14,820 netting hours (617.5 d) resulted in 255 captures of 200 green turtles. Straight-carapace length and mass ranged from 46.0–100.0 cm (mean=74.3 ±0.7 cm) and 14.5–145.0 kg (mean=61.5 ±1.7 kg), respectively. The size–frequency distribution remained stable during all years and among all capture locations. Anthropogenic-derived injuries ranging from missing flippers to boat propeller scars were present in 4% of captured turtles. Remains of 18 turtles were found at dumpsites, nine stranded turtles were encountered in the study area, and flipper tags from seven turtles were recovered. Survival was estimated at 0.58 for juveniles and 0.97 for adults using a joint live-recapture and dead-recovery model (Burnham model). Low survival among juveniles, declining annual catch per unit effort, and the presence of butchered carcasses indicated human activities continue to impact green turtles at this foraging area.

INTRODUCTION

Understanding the status of sea turtle populations at coastal foraging areas is fundamental to their conservation. In contrast to nesting beach surveys that focus on adult females, foraging area assessments can provide demographic information on a broad range of age-classes, thus giving insights about present and future population abundance trends (Chaloupka & Musick, 1997). The need for status assessments in foraging areas has been discussed (e.g. Frazer, 1992; Bjorndal, 1999) but few have been undertaken, perhaps due to the labour-intensive and expensive nature of such studies. Nonetheless, data generated from these studies will be vital for future sea turtle conservation efforts.

Green turtles, *Chelonia mydas* (Linnaeus, 1758), inhabit neritic foraging areas in tropical and subtropical regions throughout the world's oceans. Due to overexploitation of eggs and meat as a food resource and, to a lesser extent, incidental mortality relating to marine fisheries and degradation of marine and nesting habitats, green turtle populations have declined throughout the world (Groombridge & Luxmoore, 1989). Green turtles are currently listed as endangered in the World Conservation Union (IUCN) Red Data Book (Hilton-Taylor, 2000) and are included in Appendix 1 of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Despite a worldwide increase in research and conservation of green turtles, their demography in foraging areas remains poorly understood.

In the eastern Pacific Ocean, green turtles (also known as black turtles, *Chelonia mydas agassizii*) have experienced substantial declines due to human overexploitation (Cliffton et al., 1982; Figueroa et al., 1993). The annual nesting population at the primary rookery in Michoacán, México, was estimated at ∼15,000 females in the early 1970s (Cliffton et al., 1982) but has since dropped to fewer than 1000 females despite nesting beach protection ongoing since 1979 (Alvarado-Díaz et al., 2001). This decline has been largely attributed to continued threats in distant neritic habitats, particularly near Baja California (Alvarado & Figueroa, 1992; Gardner & Nichols, 2001).

Green turtles have been harvested in great numbers from coastal habitats of the Baja California peninsula for nearly a century (Agler, 1913; Caldwell, 1963; Cliffton et al., 1982). One of the most impacted areas has been Bahía de los Angeles (BLA; Figure 1) where, in 1962 for example, over 150 turtles per week were harvested during summer months and a comparable number during winter months (Caldwell, 1963). In that year alone, 186.5 metric tons of sea turtle (primarily *C. mydas*) were landed in BLA (Márquez, 1984; Figure 2). However, by the 1980s

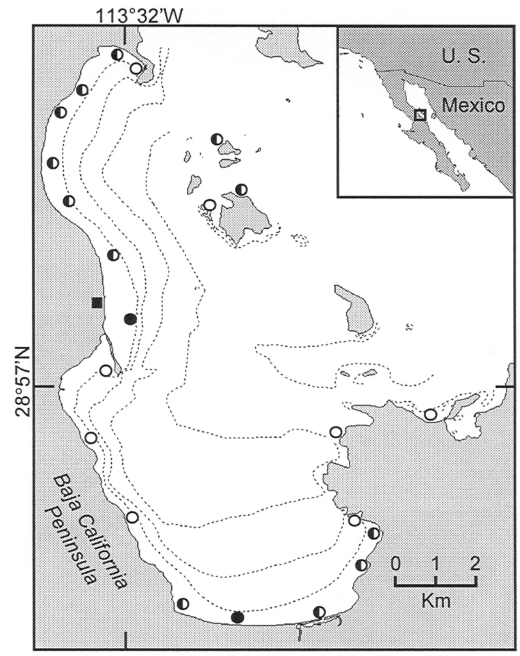

Figure 1. Map of the Bahía de los Angeles Study area along the eastern coast of the Baja California Peninsula. Hatched lines are 10-m bathymetric contours. The solid square shows the location of the Vida Silvestre—Sea Turtle Research Station. Filled circles mark the two most successful capture sites of El Bajo (northernmost) and El Rincón (southernmost); half-filled circles denote capture sites that yielded at least one turtle; and empty circles mark capture sites at which no turtles were caught.

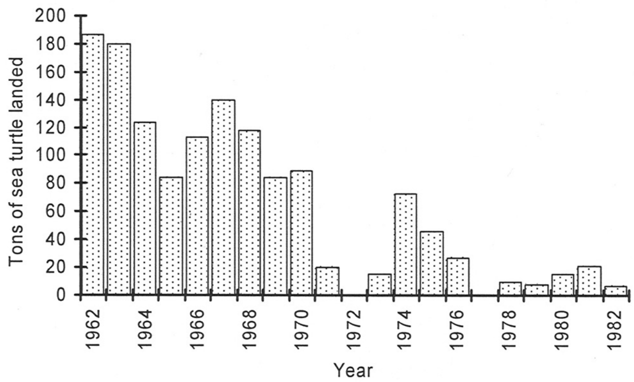

Figure 2. Summary of reported sea turtle landings (primarily *Chelonia mydas*) in Bahía de los Angeles between 1962 and 1982. The decrease in landings between 1971 and 1973 was due to implementation of fishing quotas and closures during those years. Data from Márquez (1984).

commercial harvest became increasingly difficult due to a decline of green turtle populations. In 1982, for example, only 7 metric tons of sea turtle were harvested from BLA, representing a 96% drop in annual catch rate since the 1960s (Márquez, 1984; Figure 2). The commercial turtle cooperative in BLA disbanded at the end of that year as a result (A. Resendiz, unpublished data).

Efforts to slow the decline of green turtles in neritic habitats in the eastern Pacific started in 1971 when the Mexican government legislated a series of closed seasons and quotas (Márquez, 1984). However, the widespread nature of fisheries operations and logistical difficulty of enforcing the protective regulations enabled sea turtle harvests, and ensuing population declines, to continue. Consequently, the Mexican Government banned sea turtle harvests in 1990 (Anon., 1990). Although the overall goal of this moratorium was to recover sea turtle populations, the ability to monitor its effectiveness has been hampered by a lack of demographic information from foraging areas (National Marine Fisheries Service & US Fish and Wildlife Service, 1998).

In this paper we report on the current status of green turtles at BLA and identify persisting human impacts at this coastal foraging area. This study presents the first information available on green turtle abundance trends of this region since the passage of the Mexican Government's protective legislation over a decade ago (Anon., 1990). Moreover, this is the first estimate of survival for green turtles in north-western México and provides a basis for developing a better understanding of green turtle population dynamics throughout the region.

MATERIALS AND METHODS

Study Site

The study was conducted at Bahía de los Angeles (28°58′N 113°33′W), a north-north-east-oriented bay along the eastern coast of the Baja California Peninsula, México (Figure 1). A series of 17 islands line the north-eastern portion of BLA and separate this feeding area from pelagic offshore waters of the central Gulf of California. Bahía de los Angeles is ~60 km^2 in area, and characterized by strong tidal mixing and upwelling, which support productive marine benthic communities dominated by marine algae. Habitat characteristics of BLA are further described in Seminoff (2000). A small town also named Bahía de los Angeles is located along the western shores of BLA. The local economy for this community of ~800 persons is based on artisanal fisheries, sport fishing, and nature tourism.

Turtle capture and measurement

Between June 1995 and August 2002 green turtles were captured along the shallow perimeter of BLA using entanglement nets (200×8 m, mesh size=50 cm stretched). The entanglement nets used in this study have successfully captured the smallest of post-pelagic juvenile turtles at other Gulf of California foraging areas (straight-carapace length [SCL]=35 cm), therefore we believe there is no inherent sampling bias with their use in this study. Nets were set during both day and night periods and monitored at 0.5-h to 12-h intervals. We attempted capture at 22 localities along the near-shore perimeter of the study area (Figure 1). Distance from shore and water depth of netting sites ranged from 50 to 750 m, and 2 to 27 m, respectively. Upon capture, turtles were transported 1 to 8 km to the Vida Silvestre—Sea Turtle Research Station where they were weighed, measured, and tagged. All turtles were released at the site of initial capture within 48 h.

General health of each turtle was assessed and missing flippers, large scars, and other external anomalies were noted. Straight carapace length (SCL; ±0.1 cm) from the nuchal notch to the posterior-most portion of the rear marginals was measured using a forester's caliper. To facilitate comparisons with other green turtle studies, curved carapace length (CCL; ±0.1 cm) was measured using a flexible tape from the same carapace locations. Turtles were assigned to one of two size groups delineated by the mean nesting size (MNS) of females at the Colola, Michoacan (77.3 cm SCL, N=718; Figueroa et al., 1993), considered the primary source rookery for BLA based on genetic evidence (Nichols, 2003; P. Dutton, unpublished data). Group I included all animals with SCL<MNS (i.e. probable immatures) and Group II included all animals with SCL≥MNS (i.e. probable adults). Turtles were weighed to the nearest kg using a 150-kg spring balance. Each turtle was tagged with Inconel tags (Style 681, National Band and Tag Company, Newport, Kentucky), in the first large proximal scale of each rear flipper.

A body condition index (BCI=body mass/SCL^3) was calculated to evaluate the size vs mass relationship of captured green turtles (Bjorndal et al., 2000). To eliminate the potential effects of migration on condition, BCI was determined for turtles in Group I only. Because the condition index was not correlated with SCL (Spearman's rho=0.0057, P=0.9532) and size ranges were similar among years we could compare BCI values among years (Bolger & Connolly, 1989; Bjorndal et al., 2000).

Green turtle mortality

Mortality was quantified through stranding records, counts of discarded turtle carcasses, and flipper tag returns. To quantify the occurrence of green turtle strandings (i.e. dead turtles) beach surveys were conducted along the coastal perimeter (~16 km) of the study area monthly, from June to August (1996–2002; N=21). In addition, we carried out sporadic searches of refuse dumps and fish camps for discarded carcasses. Data were recorded on each sea turtle carcass found, following Gardner & Nichols (2001). Carapace measurements followed the same procedure as that for live-captured turtles. Presence of external abnormalities was documented, necropsies were carried out when possible, and cause of death was recorded when known. To gather information on incidental mortality in local commercial fisheries activities, the return of flipper tags was solicited from local fishers during private discussions and town meetings.

Modelling survival, recapture probability, and emigration

Live-recapture and dead-recovery data were analysed for green turtles in BLA to estimate true survival, probability of recapture, and probability of emigration. Turtles were identified by their unique flipper tag numbers. Only the initial live-capture of a turtle in any given year was used in this analysis. Tag loss was not a significant concern due to the fact that we applied two flipper tags on each turtle and tags were replaced as needed after recapture. Moreover, no turtles were captured that bore scars indicative of flipper tag loss. The dead-recovery data were based on tags returned to us from fishers and on turtles that were marked during this study and later found during our stranding surveys. The survival model used here (see below) does not assume demographic closure and is therefore suitable for studies at sea turtle foraging areas, which are commonly open systems. By incorporating both live-recapture and dead-recovery data, our survival model allowed us to more accurately distinguish between mortality and permanent emigration (Burnham, 1993; Bjorndal et al., 2003).

Statistical analyses

Annual catch per unit effort (CPUE) was calculated for each year of this study (1995–2002). One unit effort is equal to one 24-h in-water set for a single 200-m net. The relationships between green turtle size (log SCL) and size predictors (year and capture site) and green turtle CI (Group I only) and CI predictors (year, capture site) were analysed statistically using general linear modelling (GLM; see Cohen et al., 2003). Regression techniques were used to characterize the relationship between SCL and CCL, and SCL and body mass. T-tests were used to compare mean size of known-fate turtles vs live-captures. Wald Chi-square analyses determined the homogeneity of the proportion of turtles in Groups I and II among years. Statistical analyses were done with JMP software (SAS Institute, Belmont, California). Mean values are followed by standard error (±SE).

True survival and probability of emigration were estimated using a Burnham-type model with logit link (joint estimation of live-recapture model and tag recovery model; Burnham, 1993; Catchpole et al., 1998) in the program MARK (White & Burnham, 1999). Model parameters were survival probability (S), recapture probability (p), fidelity probability (F), and recovery probability (r), which is the probability that a tagged turtle that dies is found and also that the tag is returned to the study team. This last probability is important because not all tags on turtles that are killed or found dead are returned. In addition to being lost in the mail, tags may not be returned because the finder does not understand the message on the tag, fears revealing the capture if the turtle was taken illegally, does not have sufficient funds to mail the tag, or is indifferent. Relative model fit was assessed using the quasi-likelihood corrected form of the Akaike information criterion (QAICc; Anderson et al., 1998). The goodness of fit (GoF) of the best model selected by QAICc was then assessed using a parametric bootstrap approach employed in MARK. Further explanation of analysis of sea turtle capture–mark–recapture studies is found in Chaloupka & Limpus (2002) and Bjorndal et al. (2003).

RESULTS

Capture effort and success

A total of 14,820 net-set hours (617.5 d) yielded 255 captures of 200 green turtles. Capture was attempted during 11 calendar months with the largest proportion of turtles landed in July (N=73; Figure 3). Captures were successful from 14 sites located throughout the shallow perimeter of BLA. The greatest capture efforts occurred directly over the marine algae pastures at El Bajo and El

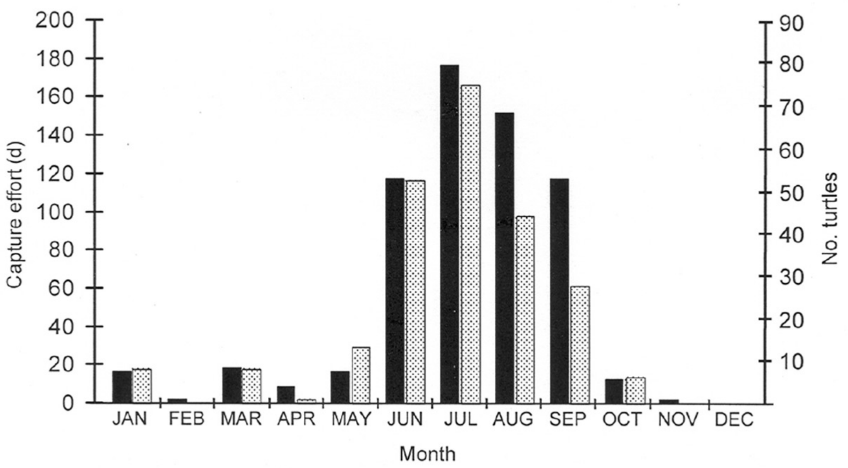

Figure 3. Monthly capture effort (filled bars) and number of captures (new captures and recaptures from previous years; shaded bars) at the Bahía de los Angeles foraging area between June 1995 and August 2002.

Rincón (Figure 1), with 6219 h and 2320 h netting effort, respectively. These sites accounted for 60% of all captures, yielding 115 captures (94 turtles) from El Bajo and 39 captures (32 turtles) from El Rincón.

Turtles were successfully recaptured from previous seasons during all but the initial year (1995) of this study. Forty-two turtles (21.0%) were recaptured on 55 occasions; 32 turtles recaptured once, eight turtles recaptured twice, one on three occasions, and one on four occasions. The distribution of recaptures among years was: 8 (1996); 3 (1997); 12 (1998); 15 (1999); 7 (2000); 5 (2001); and 5 (2002). The mean initial SCL for all recaptured turtles was 79.7 cm (SE=1.9, range=65.9–94.7 cm, N=42). The mean recapture interval was 231 d (SE=46.4, range=3–1523 d, N=55). Growth data for these recaptures are presented in Seminoff et al. (2002).

There was substantial variation in CPUE over the course of this study, ranging from 0.69 (one turtle per 34.8 h) in 1996 to 0.23 (one turtle per 104.3 h) in 2002. The annual CPUEs were highest during the earlier years of this study (1995, 1996, and 1998) and, with the exception of 1997, were lowest during the final four years of this study. The values for annual CPUE were (in descending order): 0.66 (1996); 0.65 (1998); 0.50 (1995); 0.34 (2000); 0.33 (1999); 0.27 (1997); 0.26 (2001); and 0.23 (2002).

Size of green turtles

Green turtles captured during this study ranged from 46.0–100 cm SCL (mean at first capture=74.3 ±0.7 cm, N=200; Table 1, Figure 4) and 48.5–104.3 cm CCL (mean at first capture=80.9 ±0.8 cm, N=200). A regression of SCL to CCL resulted in the linear relationship: $CCL=1.0363*SCL+2.2464$ (R^2=0.9502). The GLM model regression analysis indicates that green turtle SCL was independent of year and capture site (R^2=0.28, P=0.52). When inspected individually, neither year ($F_{7,199}$=1.99, P=0.06) nor capture site ($F_{13,199}$=0.37, P=0.98) had a significant effect on SCL. Body mass was measured for 189 turtles and ranged from 14.5–145.5 kg (mean=62.3 ±1.8 kg). A regression of SCL to body mass resulted in the curvilinear relationship: $Mass\ (kg)=2.8621e^{0.0396}$ cm SCL (R^2=0.8631).

A total of 112 green turtles were classified in Group I and 88 in Group II (Table 1). The proportion of turtles in these two groups did not vary significantly among years (Wald Chi-square=13.18, P=0.07).

Body condition

Eight (4.0%) green turtles examined during this study possessed injuries from attempted depredation and/or anthropogenic causes. Four turtles possessed injuries consistent with boat or propeller collisions, three turtles had a front flipper missing, and one turtle had large notches carved into each of the posterior-most pair of marginal scutes. Flipper loss in at least one instance was attributed to human impact: heavy gauged mono-filament fishing line was tied around the exposed front left humerus. The mono-filament line likely severed the front flipper as the turtle struggled to escape. We believe the carved marginals were modified to create grooves at which a leash could be tied. This restraint technique has been previously noted in the region (A. Resendiz, unpublished data).

Body condition was calculated for 102 turtles in Group I and ranged from 1.03 to 2.19 (mean = 1.42 ±0.015; Table 2).

Table 1. *Summary of green turtles captured at the Bahía de los Angeles foraging area between June 1995 and August 2002. Group I included all animals with straight carapace length (SCL) < mean nesting size (MNS) at the closest major rookery and Group II included all animals with SCL ≥ MNS.*

	No. captures		Size (cm)	
Year	Group I	Group II	SCL (±SE)	Range
1995	8	4	70.7 ±2.8	49.3–84.0
1996	16	22	79.8 ±1.6	61.9–100.0
1997	17	8	76.0 ±2.1	50.4–93.8
1998	17	22	77.5 ±1.7	46.0–90.7
1999	18	14	72.6 ±1.8	46.6–92.5
2000	13	8	74.6 ±2.5	50.5–95.7
2001	8	5	73.9 ±2.9	52.9–99.0
2002	15	5	71.8 ±2.3	57.5–85.5
Total	112	88	74.8 ±0.7	46.0–100.0

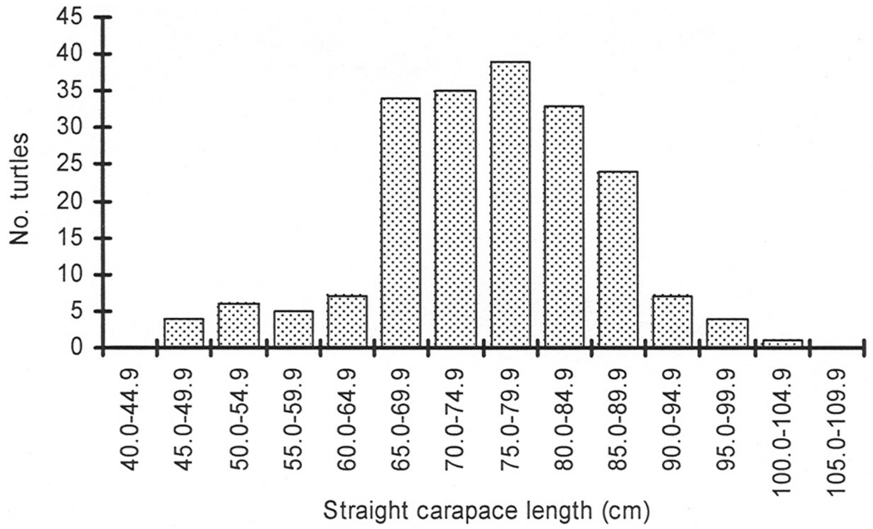

Figure 4. Histogram of straight carapace lengths of green turtles captured at the Bahía de los Angeles foraging area between 1995 and 2002 (N=200 turtles).

Table 2. *Mean annual body condition index for green turtles captured in Bahía de los Angeles between 1995 and 2002. BCI denotes body condition index (body mass/SCL^3).*

	Body condition index			
Year	BCI (*10^4)	SE (*10^6)	Range	N
1995	1.44	3.48	1.25–1.63	12
1996	1.39	4.26	1.03–1.67	14
1997	1.38	4.26	1.10–1.78	15
1998	1.46	4.26	1.30–1.66	14
1999	1.41	3.87	1.17–1.68	17
2000	1.43	5.32	1.30–1.56	9
2001	1.47	6.03	1.13–2.19	7
2002	1.42	4.26	1.23–1.69	14
Total	1.42	0.015	1.03–2.19	102

The GLM model regression analysis indicates that CI values were independent of year and capture site (R^2=0.33, P=0.12). When inspected individually neither covariate had a significant effect on CI (year $F_{7,101}$=1.15, P=0.33; capture site $F_{13,101}$=1.20, P=0.27). Condition indices did not vary significantly between injured (N=5) and non-injured turtles (N=97; t=−1.07, P=0.29) despite the fact that three of the injured turtles had flippers missing.

Stranding surveys and flipper tag recoveries

Nine green turtles (five with flipper tags and four untagged) were found stranded in the study area. Mean SCL of this group was 75.4 ±5.0 cm (range=60.0–87.0 cm) and did not differ significantly from the mean size of live-captured turtles (t=0.0088, P=0.92; flipper tagged strandings excluded from live-capture data set). No immediate cause of death was apparent, but incidental bycatch in drift- or set-nets was likely: gillnets are commonly utilized in the region for harvest of a variety of finfish species (J. Seminoff, personal observation). We saw no evidence of contact with oil or tar, no turtles had boat collision or propeller damage, and necropsies revealed that all had full stomachs at the time of death.

Seven turtle carcasses were found at dumpsites and 11 at commercial fishing camps near BLA. Overall mean SCL of these turtles was 70.8 ±3.6 cm (range=54.5–88.0 cm) and did not differ significantly from the mean size of live-captured turtles (t=1.51, P=0.22). It was not possible to determine the cause of death for these turtles; however, the fact that all had meat removed and were actively discarded suggests they were consumed by humans.

Fishermen returned flipper tags from seven turtles; all but one were confirmed to have drowned in gill nets. The mean SCL of this group was 67.3 ±2.4 cm (range=53.8–72.1 cm). The flipper tag returns (N=7) and tagged turtles found stranded in BLA (N=5) represent a known mortality of 6.5% of all turtles marked during this study.

Survival and emigration estimates

Six models were fitted to the live-recapture and dead-recovery data (Table 3). The best relative fitting model was the Burnham model (*S(g) p(g) r(.) F(.)*) (Model 1 in Table 3). Overall, this model fitted the data adequately, which was assessed using a parametric bootstrap test (N=1000, P=0.31). Model parameters included group-specific survival probabilities (S), group-specific recapture probabilities (p), constant fidelity probability (F), and constant probability that a tag was recovered and reported (r). In this model, 1-F is an estimate of permanent emigration and S is true survival, not apparent survival from a Cormack–Jolly–Seber model (which would confound permanent emigration with mortality except in a Burnham-type model). Model 2 had a QAICc which also indicated a good fit. However, it is important to note that the best model is determined by QAICc *and* AICc Weight. Whereas the AICc Weight of Model 2 was 0.402 (40% chance of being correct), that for Model 1 was 0.546 (55% of being correct), thus indicating that it was the better fitting model.

Parameter estimates for the relative best fit model (Model 1), including standard errors and 95% confidence intervals (CI), are given in Table 4. The mean survival for green turtles in Group I (i.e. probable immatures) was 0.58 (95% CI=0.36–0.78), while for turtles in Group II

Table 3. *Modelling summary for green turtles in Bahía de los Angeles based on an 8-y (1995–2002) capture–mark–recapture study. Model parameters are group specific survival probability (S), group-specific recapture probabilities (p), constant probability that tags were encountered and reported (r), and constant fidelity probability (F). Par is number of estimable parameters. QAICc is quasi likelihood corrected Akaike information criterion. AICc weight is a relative measure of the support in the data for that model compared to all other models evaluated; deviance is relative deviance; groups (2) equals two groups separated by mean nesting size at the closest major rookery; groups (1) based on all turtles pooled; const is constant. Best relative fit is shown in bold.*

Model no.	No. groups	S	*p*	*r*	*F*	Par	QIACc	AICc weight	Deviance
1	**2**	**group**	**group**	**const**	**const**	**6**	**344.1**	**0.546**	**94.4**
2	2	group	const	const	const	5	344.7	0.402	97.2
3	1	const	const	const	const	4	349.4	0.037	104.0
4	2	const	group	const	const	5	351.5	0.013	104.0
5	1	time	time	time	time	22	369.4	0.000	83.3
6	2	group×time	group×time	group×time	group×time	35	379.5	0.000	59.3

Table 4. *Estimates for true survival (S), recapture probability (ρ), probability that tags were reported (r), and fidelity (F) for green turtles in Bahía de los Angeles from 2-group Burnham model (live-recaptures and dead-recoveries; Table 3, Model 1). Group I included all animals with straight carapace length (SCL) < mean nesting size (MNS) at the closest major rookery and Group II included all animals with SCL ⩾ MNS.*

Parameter	Estimate	SE	95% CI
$S_{\text{Group I}}$	0.5848	0.1141	0.3593–0.7797
$S_{\text{Group II}}$	0.9791	0.0229	0.8391–0.9976
$\rho_{\text{Group I}}$	0.3697	0.1275	0.1671–0.6315
$\rho_{\text{Group II}}$	0.1818	0.0706	0.0806–0.3604
r	0.1174	0.0352	0.0640–0.2057
F	0.5378	0.0998	0.3461–0.7189

SE, standard error; CI, confidence interval.

(i.e. probable adults) it was 0.98 (95% CI=0.84–0.99). The mean recapture probabilities for turtles in Group I and Group II were 0.37 (95% CI=0.17–0.63) and 0.18 (95% CI=0.08–0.36), respectively. Fidelity was constant between groups (F=0.53; 95% CI=0.35–0.71). The probability that a tag was found and reported was also constant between groups, measuring 0.12 (95% CI=0.06–0.21).

DISCUSSION

Size and body condition of green turtles at Bahía de los Angeles

More than four decades have passed since Caldwell's (1962) report on the commercial green turtle fishery in Bahía de los Angeles. Despite substantial perturbations to the population in the years since, the size-range of turtles examined in the present study (46.0–100.0 cm) is strikingly similar to that found in 1962 (46.4–97.8 cm SCL, N=323; Caldwell, 1962). Considering that most sea turtles attain maturity at or near mean nesting size (Limpus et al., 1994), the size-range of green turtles captured during both periods is representative of large-immature and adult turtles.

Green turtles in the eastern Pacific Ocean shift from the oceanic juvenile phase to the neritic juvenile phase at 35 to 40 cm SCL (Nichols, 2003; J. Seminoff, unpublished data), but individuals smaller than 46 cm SCL were not recorded in near-shore habitats of BLA. The absence of small, post-pelagic green turtles may result from an ontogenic shift in habitat preference. Whereas the relatively deep, exposed waters of BLA host larger juveniles and adults (overall mean=74.3 ±0.7 cm SCL), capture data from the Infiernillo Channel, a shallow, protected foraging area ~50 km to the east show a prevalence of smaller juveniles (starting at 38.9 cm SCL, mean 59.7 ±1.4 cm; J. Seminoff, unpublished data). Similarly, along the Pacific Coast of the Baja California Peninsula, López-Mendilaharsu (2002) reported that mean SCL of turtles captured in high-energy Pacific coastal waters (67.7 ±3.1 cm) was significantly greater ($t=-2.92$, P=0.008) than turtles captured in a nearby lagoon system (55.5 ±2.8 cm).

The mean body condition index of green turtles examined in this study (mean=1.42 ±0.015) is significantly higher than that calculated (a posteriori) for green turtles examined by Caldwell (1962; BCI=1.21 ±0.023, range= 0.98−1.38; t=6.658, $P<0.001$). The difference may be due to an intrinsic population density effect similar to that described by Bjorndal et al. (2000) for juvenile green turtles in a Bahaman developmental habitat. The BLA population has been subjected to high levels of extraction since Caldwell's (1962) study and, considering the low CPUE of this study, it is likely that the current BLA population is well below local carrying capacity. The presumed vacant foraging habitats and greater per-turtle abundance of food may be conducive to better body condition. Alternatively, the lower BCI of turtles in Caldwell's (1962) study could have been a result of different captive handling protocols relative to this study or due to measurement error.

Survival, residency and emigration

The 200 green turtle capture-profiles were classified into two groups for estimation of survival probabilities. Because age was unknown, we used two life-stage classes (immatures, adults) that were delineated by the mean nesting size of females at the closest major rookery. Although a common goal of survival modelling is to determine annual values for survival for each age-class (i.e. Model 6, Table 3), given our small data set, such a model would be over-parameterized, as indicated by the high QAICc value in Table 3.

Estimates of true survival probabilities (S) and recapture probabilities (p) in Groups I and II show an inverse relationship. This suggests that when turtles are less prone to recapture, their survival increases. The reasons for variation in recapture probability are unclear. Perhaps there is a life-stage based difference in the movement patterns, with adults (Group II) having a greater tendency to make longer temporary movements out of the study area, thus lessening their chance for recapture. If mortality causes are more common within the study area, this would explain the higher survival observed in adults. Alternatively, adults may be more 'trap shy' than juveniles. If turtles surviving to adulthood have done so in part due to their predisposition for being more wary of nets, it is possible that they would be less prone to multiple capture in the nets used in this study as well as nets from local commercial fisheries. However, to our knowledge no such reports have come from other in-water sea turtle studies.

Under natural conditions, the lower survival for Group I would not be surprising due to the fact that smaller turtles are usually more susceptible to predation (Musick & Limpus, 1997). Mortality in the present study is, however, due to both natural and anthropogenic-induced factors. Although it is not possible to unequivocally separate the two, the strandings, butchered carcasses, and tag recoveries suggest that the latter is more prevalent. This is supported by the fact that our estimate of Group I survival (0.58) is notably low compared to juvenile survival reported from less impacted foraging habitats (for review see Chaloupka & Limpus (2002) and Bjorndal et al. (2003)). However, it is possible that immature survival was underestimated here due to the dispersal behaviour related to the onset of sexual maturity and ontogenic migrations (Musick & Limpus, 1997). To more adequately explore this possibility and determine if there is local dispersal from BLA, we encourage additional studies at adjacent foraging areas.

In contrast to Group I, the survival estimate for turtles in Group II (0.97) is remarkably high. Although this value is indicative of a hazard-free age-class, this result must be viewed with caution, particularly when considering the prevalence of local human impacts. The comparatively smaller sample size of Group II individuals weakens the model's power and we caution against drawing specific conclusions based on this study alone. Nonetheless, the relatively high survival for adult green turtles is consistent with high survival probabilities reported for adult loggerheads and green turtles in the southern Great Barrier Reef (Chaloupka & Limpus, 2002).

Although the possibility that turtles departed this area between captures can not be ruled out, the recapture of green turtles within years and in consecutive years and the corresponding estimate of fidelity (F) suggest that a significant proportion of the green turtles captured during this study reside in this foraging area for extended intervals. Affinity to sites with abundant food resources such as the marine algae pastures at BLA is consistent with reports from other green turtle foraging grounds (Musick & Limpus, 1997; and references therein).

Recovery potential for green turtles near Baja California

Clearly, the goal of the Mexican Government's 1990 moratorium (Anon., 1990) was to stem human impacts on sea turtles and initiate population recoveries throughout Mexico. However, although this legislation has provided a legal framework to promote conservation, it has had negligible success in BLA. Considering that nearly 500 green turtles from this feeding ground were landed over a three-week period in the summer of 1962 (Caldwell, 1962), the relatively few captures during this eight-year study (200 turtles) suggest that the local green turtle population remains substantially depleted relative to former levels. Further, three lines of evidence from this study indicate that human impacts persist: (1) direct observation of stranded and butchered carcasses; (2) a particularly low survival estimate for juvenile green turtles (0.58); and (3) a declining CPUE over the later half of this study's duration.

These results have alarming implications for the conservation of local green turtles, especially when considering their slow maturation rate. Based on growth data, a newly settling green turtle in BLA requires nearly 20 years to mature (Seminoff et al., 2002). With less than two-thirds of juveniles surviving annually, the likelihood of an individual reaching adulthood and reproducing is therefore very low. Morevoer, in addition to the increased exposure to anthropogenic threats caused by a long juvenile life-phase, delayed maturation substantially limits the overall growth potential of sea turtle populations (Crouse, 1999). Thus, even if the human impacts at BLA and its source rookery in Michoacán are halted completely, the recovery of this population will take many years.

Current evidence suggests that the status of green turtles near BLA is consistent with circumstances throughout Baja California: Nichols (2003) estimates that the annual mortality in this region (due to direct and indirect take of adults and juveniles) is between 10,000 and 35,000 turtles. It is thus apparent that on-the-ground protection efforts must be undertaken throughout Baja California to facilitate green turtle recoveries. Conservation strategies should be developed and implemented that are long-term and target all life-stages of green turtles. Efforts should be broad-based, extending from marine habitats to encompass urban centres that are often the source of demands for illegal turtle products (Nichols, 2003).

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