Monitoring and conservation of critically reduced marine turtle nesting populations: lessons from the Cayman Islands

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Abstract

Historically, nesting marine turtles were abundant in the Cayman Islands and were an integral part of the economy and culture. Today, nesting of loggerhead turtle *Caretta caretta* and green turtles *Chelonia mydas* takes place at very low levels. Hawksbill *Eretmochelys imbricata* nesting has not been recorded since 1999. We overview highly detailed monitoring data gathered over a 6-year period allowing insight into the magnitude and spatial and temporal patterns of marine turtle nesting, cost-effectiveness of monitoring such reduced populations, impacts of development on reproductive success and current threats to the recovery of the population. Nesting is diffuse and widely distributed for both nesting species on Grand and Little Cayman. Modelled nesting detection profiles for Grand Cayman show that in order to maintain data resolution, most sandy coastline must be surveyed throughout each season. However, in Little Cayman it may be possible to reduce effort. Legal take of adults and illegal take of eggs may be significantly impacting the remaining population. Surprisingly, we observed no significant correlation between density of coastal development and clutch density, adult emergence success or hatching success for either species. A significant relationship exists however, between density of coastal development and incidence of misorientation events in loggerhead hatchlings but not in green turtle hatchlings. Effective protection of known nesting habitat and the elimination of exploitation of remaining adults and eggs within the population are critical to its recovery.

Introduction

Historically, nesting turtles were abundant in the Cayman Islands (Lewis, 1940; Williams, 1995) and it has been suggested that these islands may have supported the largest rookery for the green turtle *Chelonia mydas* in the Caribbean (Groombridge, 1982; King, 1982). Intensive commercial fishing of turtles from as early as the 17th century (Jackson, 1977) effectively exhausted local populations by the early 1800s, and by 1900 it was believed that the local reproductive population had been extirpated (Parsons, 1962; Stoddart, 1980*a*; Groombridge, 1982; King, 1982). Wood & Wood (1994), however, reported 78 marine turtle nests by four species of marine turtle (green *Chelonia mydas*, loggerhead *Caretta caretta*, hawksbill *Eretmochelys imbricata* and leatherback *Dermochelys coriacea*) between 1971 and 1991, 76 being on Grand Cayman, with one on each of Little Cayman and Cayman Brac. Building upon this work, Aiken *et al.* (2001*a,b*) reported the results of the first systematic survey for nesting conducted during 1998 and 1999, identifying small populations of green and loggerhead turtles and critically low numbers of hawksbill turtles still nesting in the Cayman Islands, which have not been recorded since.

Globally, green and loggerhead turtles are classified by the IUCN as endangered (Hilton-Taylor, 2000). Throughout the Caribbean, major green turtle populations remain in Costa Rica, Venezuela, Mexico and the USA (Seminoff, 2004). Major loggerhead turtle populations in this region are found in the USA (Meylan, Schroeder & Mosier, 1995; TEWG, 2000), Mexico (Zurita, Herrera & Prezas, 1993) and Cuba (Felix Moncada, Ministry of Industrial Fisheries, Cuba, pers. comm.). The largest populations of the critically endangered hawksbill turtle (IUCN, 2006) in the Caribbean are found in Mexico (Garduno-Andrade *et al.*, 1999), with additional regionally significant rookeries in Antigua (Meylan, 1999; Richardson, Bell & Richardson, 1999), Barbados (Krueger, Horrocks & Beggs, 2003), Cuba (Moncada *et al.*, 1999), Panama (Meylan, 1999), Puerto Rico (Meylan, 1999), Turks and Caicos, US Virgin Islands (Godley *et al.*, 2004) and Venezuela (Meylan, 1999). In addition to these rookeries, many islands in the region, including the Cayman Islands, host small, presumedly depleted nesting populations of one or more species.

A wide suite of anthropogenic threats has been identified for marine turtles in their nesting habitat across their range (Groombridge & Luxmoore, 1989; Lutcavage *et al.*, 1997;

Witherington, 1999). This is especially true for relatively highly developed island states such as the Cayman Islands, where nesting persists in close proximity to urbanized habitation and facilities for mass tourism. Tourism in the Cayman Islands accounted for an average of 3 26 000 air arrivals and 1.3 million cruise ship arrivals annually during the period of this study (1999–2003; Cayman Islands Government, unpubl. data), and is accompanied by escalating levels of coastal development and an ever-expanding water sports industry (Ebanks & Bush, 1990; Tratalos & Austin, 2001).

Here, we review the results of 6 years of comprehensive monitoring of marine turtle nesting sites in the Cayman Islands. Data are analysed to gain an understanding of spatial and temporal patterns of nesting. Difficulties in attempting to spatially restrict monitoring of such remnant populations to promote cost-effectiveness are evaluated using the Cayman Islands as a case study. Density of coastal development is quantified, and data are analysed to determine anthropogenic impacts on reproductive success. We analyse potential threats to the remaining populations and suggest priority steps that should be taken to promote the conservation and recovery of marine turtle nesting populations in the Cayman Islands.

Study site

The Cayman Islands are located in the Caribbean Sea and consist of three islands: Grand Cayman 19°21N, 81°17W; Little Cayman 19°43N, 80°03W; Cayman Brac 19°43N, 79°51W (Fig. 1). On the basis of beach morphology and composition, and information from historical records and anecdotal reports (Diez & Ottenwalder, 1999), the following portions of the coastline were identified as suitable for marine turtle nesting: Grand Cayman, 27 beaches, covering 35 km of a total of 127 km of coastline; Little Cayman, 16 beaches, covering 18 km of a total 37 km of coastline; and Cayman Brac, seven beaches, covering 3 km of a total 41 km of coastline.
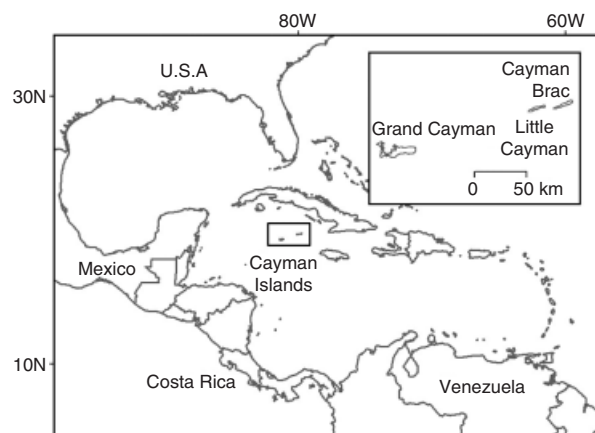

Figure 1 Map showing location of the Cayman Islands within the Caribbean region.

Methods

Standardized monitoring was carried out on Grand Cayman in each of the years from 1999 to 2003 and on Little Cayman in 1998 and from 2000 to 2003. In Grand Cayman, all beaches considered suitable for marine turtle nesting based on the initial assessment described above were surveyed on foot between 5 and 11 AM twice weekly throughout the entire nesting season (May–September) to record signs of recent marine turtle nesting activity. Little Cayman survey frequency varied depending on the available resources and ranged from one to two times per week; however, survey effort did not vary systematically among years. Standardized monitoring in Cayman Brac has not taken place and only public reports of nesting have been recorded. The sandy coastline was typically divided into subsections where geographic features provided natural breaks as follows: Grand Cayman (29 sections: mean length ± SD = 1.19 ± 1.14 km; range 0.04–3.44 km), Little Cayman (16 sections: 1.15 ± 1.04 km; 0.09–3.75 km).

After initial identification of nests via observation of known nest morphology or investigation into any disturbance of the sand not obviously caused by human traffic, clutches were located and fixed by triangulation using 100 m survey tape and markers along the beach ridge. A numbered plastic tag was placed vertically in the sand 50 cm inland of the nest to aid in clutch identification. Adult nesting success was calculated as the proportion of emergences that resulted in clutch deposition (Miller, 1999). 'Potential nests' were recorded where clutches were not located; where sand disturbance indicated oviposition may have occurred, an approximate location was fixed by triangulation. After 45 days, all nests and potential nest sites were monitored on alternate days for signs of hatchling emergence, identified by a depression in the sand and/or hatchling tracks coming from the nest site. Potential nests were recorded as nests only when a clutch (eggs, shells and/or live or dead hatchlings) was identified after hatching. Nests were excavated upon observation of completed or near-complete hatchling emergence, and clutch size and hatching success were ascertained according to the methodology of Aiken *et al.* (2001*b*) and Miller (1999).

Where possible, species identification was made from track and nest morphology (Schroeder & Murphy, 1999). Occasionally, due to varying factors such as survey intensity, weather conditions, levels of pedestrian traffic on beaches or illegal take of eggs, species could not be reliably assigned. In these cases, if no hatchlings or embryos were found in the nest at the time of excavation, species were recorded as unidentified. If no hatchling emergence was observed after 70 days of incubation, nests were excavated. Misorientation events were recorded when hatchling tracks led away from the sea and/or dehydrated or dead hatchlings were found inland of the nest site.

The Grand Cayman coastline, although highly variable among different coastal sections, is much more highly developed than Little Cayman or Cayman Brac. Beachfront development may impact marine turtle nesting by providing foundation for increased lighting and human traffic.

Development density around the Grand Cayman coastline (calculated as the total seaward facing distance of all building footprints within 60 m of the high water mark) was determined using ArcMap software and locally developed data layers as an index of development. To account for multi-storey buildings, housing density per km was weighted by the number of floors present. Although this may be considered simplistic, it gives a clear indication of the current state of 'development' in terms of the level of coastal building density at the time of writing. Data from Little Cayman and Cayman Brac were not included in this part of analysis. Development on Little Cayman is at a very low level and has not been quantified in this study, and as yet the knowledge of nesting in Cayman Brac is rudimentary.

Results

Nesting biology

Magnitude of nesting

Table 1 presents the total annual number of clutches per species on each of the three islands in the seasons they were surveyed. On Grand and Little Cayman between 2000 and 2003, the mean annual ($\pm$ SD, range) number of clutches laid was 26 ($\pm$ 19.0, 5–51) for green turtles and 26 ($\pm$ 9.8, 13–35) for loggerhead turtles. On the basis of these numbers and using the methodology for green turtles outlined by Seminoff (2004), which estimates that each female lays on average three clutches every 2–3 years, nesting populations are likely to number *c*. 17–26 females for both green and loggerhead turtles. Some hawksbill nesting has been recorded on Grand and Little Cayman (two nests in Little Cayman in 1998, two nests in Grand Cayman in 1999), but none has been recorded since 1999. An average of 4.2 clutches in each year was not assigned to species (range 1–11). On Cayman Brac, Aiken *et al.* (2001*a,b*) recorded one unidentified nest in 1999, and 12 loggerhead nests were recorded in 2003 with anecdotal evidence suggesting that a further four green turtle nests were laid in 2002. For green turtles, annual fluctuation in nesting numbers is substantial, although the interannual variability, as described by the coefficient of variation (CV = SD/mean) of nesting numbers, was within the species range (Cayman Islands CV = 0.94; global range 0.41–1.08; Broderick, Godley & Hays, 2001), while for loggerhead turtles, variability was slightly higher than the previously

documented global range (Cayman Islands CV = 0.38; global range 0.13–0.35; Broderick *et al.*, 2001).

Seasonality of nesting

The marine turtle nesting season in the Cayman Islands typically extends from May to September (absolute range: 30 April–11 September; Fig. 2a). Generally, loggerhead nesting begins in the first week of May and usually continues through to the first week of August. Green turtle nesting typically begins in late June and continues no later than to the end of September, although nesting events have been recorded between 29 May and 5 October (Fig. 2a). Two hawksbill nests were recorded in each of July 1998 and August 1999 (Aiken *et al.*, 2001*b*) and a leatherback nonnesting emergence was recorded in Cayman Brac in July 2003.

It is likely that seasonality of nesting is tied to climatic factors leading to suitable nest construction and incubation environments. There is a marked, although limited, seasonal pattern in air temperatures and little air temperature fluctuation during the reproductive season (Fig. 2a; range of mean temperatures: 28.1–30.1 °C). Marine turtle nesting and incubation occur in the warmest months. Rainfall patterns (Fig. 2b) show much higher levels of interannual variability and therefore a less consistent seasonal pattern. Marine turtle reproduction appears less tightly coupled to patterns in rainfall, although nesting typically occurs during the wettest months (Fig. 2b), which are also the warmest. Irrespective of the causality, the fact that the two species have different, albeit overlapping, seasons means choosing an index temporal window for monitoring efforts as an indicator of nesting effort for both species.

Spatial distribution of nesting

Figure 3 shows the spatial distribution of nesting on the three islands. Nesting is diffuse and widely distributed on Grand (Fig. 3a) and Little (Fig. 3b) Cayman, and has been confirmed on only one beach to date for Cayman Brac (Fig. 3c). Although many sites on Grand and Little Cayman host mixed species nesting, a ranked analysis of mean annual clutch density per beach for green and loggerhead turtles on Grand Cayman between 1999 and 2003 showed no significant correlation (Spearman's rank: R_s = 0.07, $P \geq 0.05$). In contrast, a statistically significant correlation was evident in

Table 1 Magnitude of Marine turtles nesting (green turtle, *Chelonia mydas*, loggerhead turtle, *Caretta caretta*, hawksbill turtle, *Eretmochelys imbricata*) in Grand Cayman and Little Cayman, 1998-2003

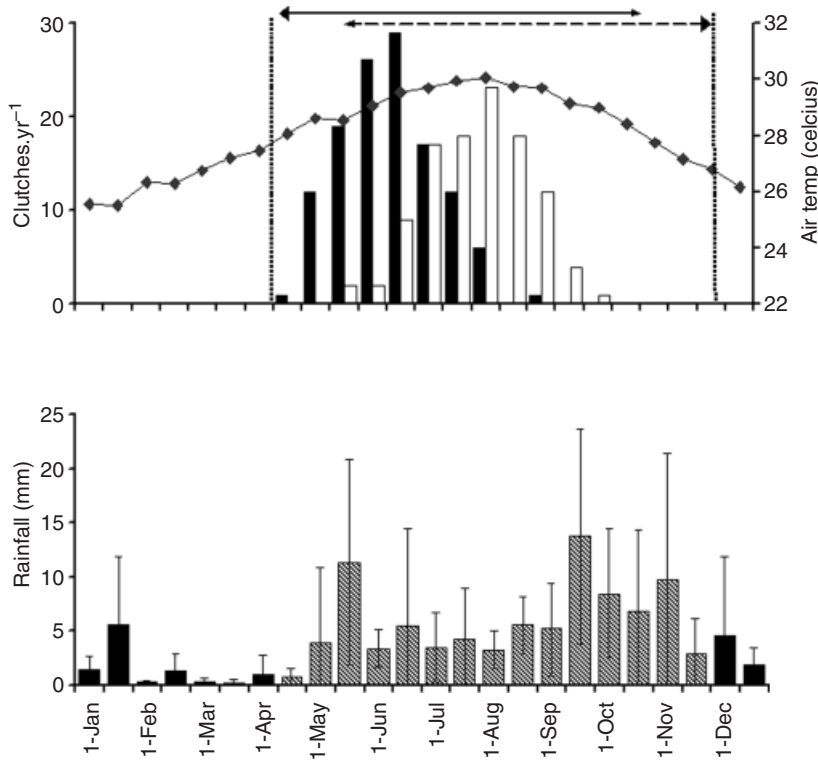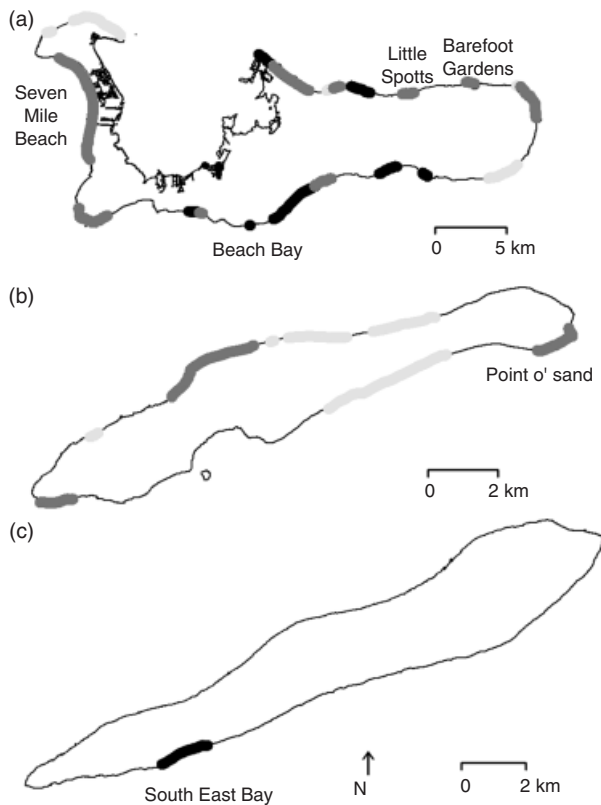|	Grand Cayman					Little Cayman				
	1999	2000	2001	2002	2003	1998	2000	2001	2002	2003
Chelonia mydas	1	18	2	43	18	9	9	3	8	4
Caretta caretta	18	25	27	12	31	0	0	5	1	4
Eretmochelys imbricata	2	0	0	0	0	2	0	0	0	0
Unidentified	2	2	1	9	1	4	2	0	2	0
Total	23	45	30	64	50	15	11	8	11	8

Figure 2 Temporal distribution of (a) loggerhead turtle *Caretta caretta* (black bars) (1999–2003), green turtle *Chelonia mydas* (white bars) nesting (1998–2003), and 5 year (1999–2003) mean air temperature (°C) (black diamonds). The combined extent of the incubation period for the two species is indicated by vertical dashed lines. Loggerhead turtle incubation period is demonstrated by the solid arrow, green turtle incubation period by the dashed arrow. (b) Mean annual rainfall (mm) in the Cayman Islands (1999–2003). Range bars show the max and min recorded during each 15 day bin over the 5 years. Shaded bars indicate the extent of the period during which clutch incubation occurs.

the nesting sites for the two species in Little Cayman during 1998 and from 2000 to 2003 ($R_s = 0.58$, $P \leq 0.05$).

The small number of nesting turtles and the spatially diffuse nature of nesting poses particular difficulties with regard to reducing monitoring effort. Across all years, no nests were recorded for either species on 2.5% of the total distance surveyed on Grand Cayman, but on 23% of the total distance surveyed on Little Cayman. On the basis of our 5 years of data, for each of Grand (Fig. 4 a–c) and Little (Fig. 4d–f) Cayman we retrospectively modelled the proportions of clutches of loggerhead, green and both turtle species combined, which are likely to be detected monitoring only the single most prolific beach, followed by the two most prolific beaches, and so on. Modelled distance covered and proportion of clutches detected are presented cumulatively (Fig. 4a–f). The range bars represent interannual variability in modelled results and are ultimately indicators of reliability of mean estimates.

It is clear that there is no simple solution to the large amount of effort needed to monitor a reduced, yet spatially dispersed, colony. These profiles (Fig. 4) show that the variability of estimates using a few beaches provides unreli-

Figure 3 Spatial distribution of all nesting recorded in the Cayman islands (a) Grand Cayman 1999–2003, (b) Little Cayman 1998, 2000–2003, (c) Cayman Brac (2003). Black lines, loggerhead turtle only nesting beaches; dark grey lines, mixed species nesting branches; pale grey lines, green turtle only nesting beaches.
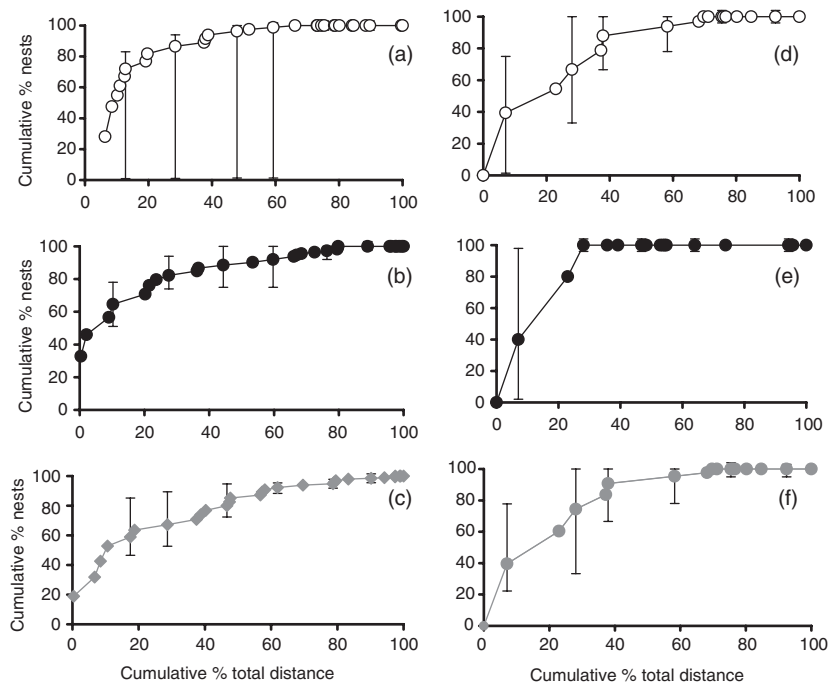
Figure 4 Cumulative percentage of clutches recorded per year vs. distance surveyed on Grand Cayman (1999–2003). (a) *Chelonia mydas*, (b) *Caretta caretta*, (c) *Ch. mydas* and *Ca. caretta* combined, and on Little Cayman (1998, 2000–2003), (d) *Ch. mydas*, (e) *Ca. caretta*, (f) *Ca. caretta* and *Ch. mydas* combined. Range bars placed at 15%, 30%, 45%, 60%, 75% and 90% of the total distance show range of nests detected at each per cent of the total distance covered during the study period, and indicate the reliability of the mean estimate.

able indices of total nesting by either species or both species combined. Beaches were ranked according to total clutch density over the study period and thus analyses are spatially consistent in order that they may inform management with respect to monitoring effort. On Grand Cayman, surveying must include the top ranking (in terms of total nesting density) 60% of the total beach for loggerhead nesting (Fig. 4a) to have detected between 75 and 100% (mean: 91%) of all loggerhead turtle nests, and surveying 80% of the total beach distance would have detected 100% of loggerhead nests in all years. For green turtles (Fig. 4b), surveying the top ranking 60% of the total distance would have detected on average 80% of nests, although interannual variability means that this would have ranged from 0 to 100%. This extensive range is due to high levels of interannual variability in the numbers of nests laid ($n = 1$–43) by this species on Grand Cayman. Available data show, however, that in any given year all green turtle nests would have been detected by covering just 66% of the total distance.

As green and loggerhead turtles appear to have both overlapping and discrete site preferences, to detect all nests of both species on Grand Cayman (Fig. 4c), 98% of all beaches would have had to be surveyed every year. Surveying the top ranking 60% of nesting habitat based on a joint species ranking would have allowed detection of 88–95% (mean: 92%) of all nests, a value that may be considered acceptable in resource-restricted areas.

On Little Cayman, if only loggerhead turtles were targeted (Fig. 4d), surveying just 30% of the total distance would have detected 100% of all nests; if only green turtles were targeted (Fig. 4e), surveying 60% of the total distance

would have detected between 78 and 100% of all nests (mean: 94%). As noted above, on Little Cayman nesting densities of the two species are correlated. If nests of both species were targeted (Fig. 4f), surveying just 60% of the distance would still have captured between 78 and 100% of all nests (mean: 95%). Covering 70% of the total distance would have captured 100% of all nests recorded on this island over five seasons.

Anthropogenic threats

As an integral part of nesting beach monitoring, efforts were made to assess the potential anthropogenic threats affecting the turtles in this habitat.

Legal take of adults

Legal take of adults of hawksbill, green and loggerhead turtles occurs within the framework of a highly regulated fishery that allows up to 84 adults in total to be removed from the population in each year; see Bell *et al.* (in press) for a review.

Illegal take of eggs and adults

Nine clutches were illegally removed from beaches in the Cayman Islands between 2000 and 2003 (four green turtle nests and five unidentified species) and none in 1998 or in 1999. This represents 3.2% of all nests in general and at least 4% of all green turtle nests recorded during this period. No cases of confirmed adult take on the beach were recorded although one attempt was foiled during routine beach monitoring for this project.

Coastal development

Given the relatively highly developed nature of some areas of the Grand Cayman coastal zone [mean $\pm$ SD building density (km frontage km^{-1}) = 0.46 ± 0.81, range 0–4.32, $n = 29$ beach sections] the possible influence of the density of coastal development on the magnitude and quality of nesting is worthy of investigation. There was no significant correlation between development density on beach segments and mean clutch density for loggerhead turtles ($R_s = 0.17$, $P > 0.05$), green turtles ($R_s = 0.68$, $P > 0.05$) or both species combined ($R_s = 0.01$, $P > 0.05$).

Some beach sections simply may not be suitable for nesting, irrespective of development levels or the apparent presence of suitable characteristics. All beaches recorded with zero nesting had development values of less than 0.5 km frontage km^{-1}. When these beaches are excluded from this analysis, Grand Cayman still showed no significant correlation between level of development (km frontage km^{-1}) and mean clutch density for loggerhead turtles ($R_s = -0.35$, $P > 0.05$), green turtles ($R_s = -0.32$, $P > 0.05$) or both species combined ($R_s = -0.267$, $P > 0.05$).

It is possible that female turtles may be more easily disturbed on developed beaches, thus lowering the 'adult emergence success' (Miller, 1999). There was, however, no significant relationship between development density (km frontage km^{-1}) and adult emergence success on all of Grand Cayman's beaches (2000–2003) that are used by loggerhead turtles ($R_s = 0.29$, $P > 0.05$), green turtles ($R_s = -0.24$, $P > 0.05$) or turtles of both species combined ($R_s = 0.12$, $P > 0.05$). Furthermore, there was no significant relationship between development density (km frontage km^{-1}) on beaches where nesting occurred and hatching success on Grand Cayman's beaches for loggerhead turtles ($R_s = 0.25$, $P > 0.05$), green turtles ($R_s = 0.26$, $P > 0.05$) or for both species combined ($R_s = 0.28$, $P > 0.05$).

Given that development is likely to lead to increased lighting, the potential relationship between development and hatchling misorientation events is worthy of investigation. There was a statistically significant relationship between level of development (km frontage km^{-1}) and proportion of loggerhead ($R_s = 0.76$, $P < 0.05$) hatchlings misoriented, but the same relationship was not significant for green turtle ($R_s = 0.18$, $P > 0.05$) hatchlings misoriented between 2000 and 2003. There is a higher incidence of misorientation in loggerhead hatchlings (6% of all recorded nests) than in green hatchlings (2% of all recorded nests).

Discussion

Data from mainland USA (Dodd, 1982; Meylan *et al.*, 1995), Hawaii (Balazs & Chaloupka, 2004), Costa Rica (Bjorndal *et al.*, 1999; Troeng & Rankin, 2005) and Ascension Island (Broderick *et al.*, 2006) show significant increases in green turtle nesting trends in the last three decades, suggesting that recovery of reproductive marine turtle populations may be possible where appropriate conservation, regulation and education strategies are applied. Our data have been collected

rigorously and although our estimates of the number of clutches per season are likely conservative, we feel it is a highly reliable index not likely to significantly underestimate the magnitude of nesting. Nesting hawksbills were recorded in 1998 and 1999 but have not been recorded since and the status of this population remains uncertain. However, it should be noted that nesting in this species may be particularly hard to detect due to nest morphology and typical location coupled with survey frequency and duration. We have shown that the Cayman Islands support populations of nesting green and loggerhead turtles, which, although critically low, do not appear to be decreasing. Notwithstanding, given that by 1900 nesting turtles in the Cayman Islands were considered extirpated (Stoddart, 1980*b*; Groombridge, 1982; King, 1982), it is possible that these data can be interpreted optimistically. Continued monitoring will allow population trends to be fully explored. Additionally, adults from the Cayman Islands Turtle Farm headstarting project are now beginning to contribute to current reproductive green turtle populations in the Cayman Islands (Aiken *et al.*, 2001*b*; Bell *et al.*, 2005), although the level of such a contribution is to date unassessed and discussion may therefore be considered premature.

Analysis of spatial patterns of nesting is of critical importance to conservation and management practices. Nest site selection and nesting success are influenced by a variety of marine and terrestrial factors (see Weishampel *et al.*, 2003 for a review). The diffuse nature of nesting and the fact that the two main nesting species appear to have differential spatial nesting preferences on Grand Cayman raise particular challenges for monitoring this population and the potential for designating index nesting beaches for long-term monitoring of population status with minimal field effort. Modelled detection profiles show that regardless of the study species, and particularly in reduced populations where nesting is widely dispersed, a large proportion of sandy coastline must be surveyed every year in order to detect all nests. This is unfortunate news for managers as it precludes the potential for developing a less resource-intensive monitoring system based on index beaches. However, in Little Cayman there is a significant positive correlation between sites of high nesting density for each species. These data show, therefore, that in cases where resources are critically low, it may be possible after an initial multi-year assessment such as ours to focus efforts on known areas of relatively high intensity. These findings have important implications for monitoring projects worldwide as new less resource-intensive approaches may be developed without negatively impacting data resolution.

It may be possible to curtail the temporal coverage of beaches to certain months and/or surveying every few years, but these strategies add additional variance into the monitoring data that further reduce the power to detect trends within the context of profound levels of interannual variation in nesting numbers (Broderick *et al.*, 2001) and very low numbers. Although intensive beach monitoring raises resource constraints to wildlife managers, it also provides opportunities to carry out surveillance regarding other anthropogenic threats such as illegal take. We found that

although eggs are still illegally excavated and taken for consumption on both Grand and Little Cayman, and although it is likely that this also takes place in Cayman Brac, it is felt that these events are not part of an organized activity but occur on an ad hoc basis when nests are found by chance or by word of mouth.

Additionally, high-resolution monitoring gives the opportunity of carrying out a correlative analysis, albeit without control, regarding nesting parameters and level of coastal development density. Perhaps, surprisingly, these results show that green and loggerhead nesting is not significantly concentrated in areas of either high or low development density on Grand Cayman; however, it should be noted that the methodology used to quantify development in this case is simplistic and may not provide suitable resolution on all parameters, although it gives a useful indication of the overall status of building density on Grand Cayman's coastline. One would expect higher levels of nesting to occur in lesser developed areas, consequently subject to lower levels of light (Witherington, 1992; Witherington & Martin, 1996; Salmon, 2002), pedestrian traffic, human presence and other external interference. Wood & Wood (1994) suggested that the construction of condominiums on many of Cayman's beaches had resulted in infrequent nesting, although the lack of correlation between coastal development and clutch density, emergence success or hatch success suggests that development has not yet systematically impacted or degraded nesting habitat.

It is possible, however, that the best potential nesting beaches may also be those that are now the most developed, thus reducing their quality to some but not all nesting females. Furthermore, small sample sizes must be given consideration before inferences can be made regarding the spatial distribution of nesting with reference to levels of development on beaches around Grand Cayman. As the numbers of nesting females are so low, and it is known that each female may nest more than once in each season (Miller, 1997), the loss or addition of one female nesting in one area in each year can significantly impact results. Irrespective of the effect of development on nesting, it is clear that photo-pollution is detrimental, with loggerhead hatchling misorientation being more prevalent on more developed beaches, which are likely to exhibit higher levels of artificial light. The higher incidence of misorientation events in loggerhead turtles over green turtles is not readily explained. It may be possible that species-specific factors, such as differential sensitivity to light waves, time of hatchling emergence, or moon phase or variation in levels of directed lighting at each site over the season, may influence susceptibility to misorientation (see Witherington & Bjorndal, 1991 for a review).

In 6 years of intensive monitoring of Cayman's reproductive population of marine turtles by the Cayman Islands' Department of Environment, a baseline for future monitoring and a foundation for conservation efforts has been established. Key nesting beaches have been identified and threats to adults and hatchlings documented and, where possible, minimized. Current measures for conservation in Cayman include protection of clutches from illegal take of eggs, steps to help minimize hatchling misorientation resultant from photo-pollution, and extensive public awareness and education programmes. Although not currently planned, the establishment of protected areas (Mortimer, 2000) at known marine turtle nesting beaches in the Cayman Islands, in addition to current protection offered by the current Marine Parks System and Turtle Protection Regulations (Cayman Islands Government, 1996), will be critical to the creation of a comprehensive framework for the protection of the marine turtle reproductive population. While neither cessation nor restriction of development in these areas is in harmony with current government policy, the establishment of regulations for current and future development that enhance and protect marine turtle nesting habitat (see Witherington & Martin, 1996 for a review) would greatly enhance future potential for the reestablishment and conservation of marine turtles in the Cayman Islands. Additionally, it is imperative that steps are taken to curtail the direct take of adults in the legal fishery in Cayman (Bell *et al.*, in press).

Acknowledgements

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