BIOLOGY AND CONSERVATION OF MARINE TURTLE NESTING
IN THE DOMINICAN REPUBLIC

Tesis Doctoral por: Ohiana Revuelta Avin
Directores: Jesús Tomás Aguirre / Juan Antonio Raga Esteve

Valencia, mayo 2014

VNIVERSITAT DE VALÈNCIA

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PROGRAMA DE DOCTORADO
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D. Jesús Tomás Aguirre, Profesor Ayudante Doctor del departamento de Zoología de la Facultad de Ciencias Biológicas de la Universitat de València y **D. Juan Antonio Raga Esteve**, Catedrático del departamento de Zoología de la Facultad de Ciencias Biológicas de la Universitat de València,

CERTIFICAN que **Dª Ohiana Revuelta Avin** ha realizado bajo nuestra dirección, y con el mayor aprovechamiento, el trabajo de investigación recogido en esta memoria, y que lleva por título "**BIOLOGY AND CONSERVATION OF MARINE TURTLE NESTING IN THE DOMINICAN REPUBLIC**", para optar al grado de Doctora en Ciencias Biológicas.

Y para que así conste, en cumplimiento de la legislación vigente, expedimos el presente certificado en Paterna, a de de 2014.

Firmado: Jesús Tomás Aguirre Firmado: Juan Antonio Raga Esteve

A Héctor

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SCIENTIFIC PUBLICATIONS FROM THE PRESENT PhD THESIS

The results of this thesis have been published, accepted or sending for its publication in the following scientific journals:

- **Ohiana Revuelta**, Yolanda M León, Pablo Feliz, Brendan J Godley, Juan A Raga and Jesús Tomás (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. **Oryx, 46, 348-358**.
- **Ohiana Revuelta**, Yolanda M León, Francisco J Aznar, Juan A Raga and Jesús Tomás (2013) Running against time: conservation of the remaining hawksbill turtle (*Eretmochelys imbricata*) nesting population in the Dominican Republic. **Journal of the Marine Biological Association of the United Kingdom, 93, 1133-1140**.
- **Ohiana Revuelta**, Yolanda M León, Annette C Broderick, Pablo Feliz, Brendan J Godley, Juan A Balbuena, Andrea Mason, Kate Poulton, Stefania Savoré, Juan A Raga and Jesús Tomás. Assessing the efficacy of direct conservation interventions: Clutch protection of the leatherback marine turtle in the Dominican Republic. **Oryx** (***In press***).
- **Ohiana Revuelta**, Brendan J Godley, Yolanda M León, Annette C Broderick, Pablo Feliz, Juan A Balbuena, Juan A Raga and Jesús Tomás. The value of endangered species in protected areas at risk: the case of the leatherback turtle in the Dominican Republic. **Biodiversity and Conservation, doi:10.1007/s10531-014-0682-x.**).
- **Ohiana Revuelta**, Lucy Hawkes, Yolanda M León, Brendan J Godley, Juan A Raga and Jesús Tomás. Evaluating the importance of Marine Protected Areas for the conservation of hawksbill turtles (*Eretmochelys imbricata*) nesting in the Dominican Republic. **Endangered Species Research** (**Submitted**).

OTHER PUBLICATIONS RELATED TO THE PRESENT PhD THESIS

During the course of these studies I have also participated in the following publications within the frame of the marine turtle conservation project in the Dominican Republic:

- Jesús Tomás, Brendan J Godley, Yolanda M León, **Ohiana Revuelta** and Juan A Raga (2011) Sea turtle conservation in the Dominican Republic: The importance of National Parks. **Testudo, 7, 32-38**.
- Lucy A Hawkes, Jesús Tomás, **Ohiana Revuelta**, Yolanda M León, Janice M Blumenthal, Annette C Broderick, Martin Fish, Juan A Raga, Mathew J Witt and Brendan J Godley (2012) Migratory patterns in hawksbill turtles described by satellite tracking. **Marine Ecology Progress Series, 461, 223-232**.
- Carlos Carreras, Brendan J Godley, Yolanda M León, Lucy A Hawkes, **Ohiana Revuelta**, Juan A Raga and Jesús Tomás (2013) Contextualising the last survivors: population structure of marine turtles in the Dominican Republic. **PLoS ONE, e66037, doi:10.1371/journal.pone.0066037.**

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SUMMARY

SUMMARY

The loss of biodiversity has become one of the most pressing issues, which has lead to a growing global concern about the status of the biological resources on which human life depends. In recent decades there has been a decrease in individual populations of many species. In the Caribbean, marine turtle's nesting rookeries have been reduced considerably, mainly due to human exploitation. A number of rookeries in this region have been studied for several decades and their status is well documented; however, many other Caribbean rookeries remain poorly described. The Dominican Republic (DR) is an area where information on marine turtle nesting activity is scarce and outdated. Surveys in the 1970s and 1980s constituted the main reference on the status of marine turtles in the DR; although more recent reports indicate that these species are under continuous threats. The lack of comprehensive studies and recent information, coupled with the evident threatened status of marine turtles in the country demanded an updated assessment to help target effective conservation actions.

The present PhD study aims to identify the main marine turtles nesting rookeries in DR, describing the current spatio-temporal patterns of nesting, and assessing the likely impact of the current threats to these nesting stocks, based on a period of 5 years (2006 - 2010) of systematic survey within a conservation project. To determine the reproductive success of *Dermochelys coriacea* and *Eretmochelys imbricata* in their remnant nesting rookeries in the country: the beaches of The Jaragua National Park (JNP, SW DR) and Saona Island (in Del Este National Park: DENP, SE DR) respectively. To face egg take by local people, the project carried out artificial incubation of a number of clutches as a conservation measure at both sites. Differences in hatching and emergence success, and sex ratio between artificially and *in situ* incubated clutches were determined to evaluate the artificial incubation programme. High risk of habitat loss was detected on the western beaches of the JNP; hence, the reproductive value of the area for leatherback turtle was determined to reinforce conservation at site. Moreover, the factors affecting hatching success were studied to ascertain what sites can be best candidates to host relocated clutches in case of habitat loss due to development plans. Another aspect of this study focusses on the investigation of the habitat-use patterns of hawksbill turtles during their internesting and foraging periods, identifying core-use areas and comparing them with current marine protected areas in the Caribbean.

The study led to the following findings and conclusions:

For the first time in recent decades we have documented the presence of three marine turtle species (leatherback, hawksbill and green turtle [*Chelonia mydas*] still nesting in the DR; although the third species was found nesting in very low numbers. Nesting is concentrated in the protected areas of JNP and Saona Island, with nesting outside these areas being relatively scarce. The JNP consistently had the highest number of clutches per year of leatherback turtles (mean 126.4 ± SD 74.1, range 17-210), with a total of 632 clutches recorded during 2006-2010. However, sporadic nesting of hawksbill and green turtles were also recorded. Saona Island hosted the ma-

jority of recorded hawksbill clutches (mean 100 ± SD 8.4 per year, range 93-111, n = 400) and green turtle clutches (mean 9.2 ± SD 6.2, range 1-15, n = 37). Sporadic nesting of leatherbacks and hawksbills was also recorded in 14 other sites in the country. According to the study of clutches, mean clutch sizes (yolked eggs) for leatherback turtle in JNP varied from 67.5 to 75.1 eggs (n = 315 clutches). Mean clutch sizes of hawksbill in Saona Island varied from 125.7 to 139.5 eggs (n = 179 clutches). The leatherback turtle nesting season extends from March to August, with a peak in May, which is consistent with reports for neighbouring nesting populations. The seasonality of hawksbill turtle nesting is similar to nearby rookeries on Antigua, Barbados and Mona Island, Puerto Rico, with a high nesting season extending from June to November. Sizes recorded from hawksbills and green turtles nesting in DR were within the normal range of sizes for both species according to literature. The mean size of leatherback turtles recorded in DR (Curved Carapace Length = 147.4 cm) is lower than the global mean, which is normally distributed around 155 cm. This could be the sign of a recovering population with a high proportion of neophyte females.

Comparison of these results with earlier reports indicates that a profound decline appears to have taken place in the last thirty years. Illegal take of eggs was identified as the main threat to marine turtles, particularly on the eastern beaches of JNP. The present study highlights the need for adequate protection and management of these areas for marine turtle conservation in the DR, since the country seemed to be an important marine turtle nesting area in the past.

In order to face threats, particularly egg take, an official program of artificial incubation was established to protect turtle clutches at JNP and Saona Island. In the JNP, clutches of the western and eastern beaches of the park (called WB and EB respectively), were incubated in hermetic boxes stored in Park rangers' huts located in both places. The present study included the assessment of efficiency of this conservation measure through the investigation of how artificial incubation might be influencing the hatching success of clutches and the resultant sex ratios.

A total of 109 leatherback clutches laid over the study period (2008-2009) were studied: 35 incubated artificially in the EB), 31 artificially incubated clutches in the WB) and 43 clutches incubated *in situ* at western beaches. The results revealed that the incubation method significantly influenced hatching success of clutches. *In situ* clutches had greater hatching success than those artificially incubated in WB and in EB. The incubation method also had an effect on the number of early stage dead embryos. The number of late stage dead embryos in a clutch was also significantly affected by the incubation method. Artificially incubated clutches in EB and WB had more late stage dead embryos than in *in situ* clutches. On the other hand, the incubation method significantly influenced incubation duration. Clutches artificially incubated in EB had longer incubation duration than *in situ* clutches and clutches artificially incubated in WB. Mean incubation temperatures were significantly lower in artificially incubated clutches at EB when compared to artificially incubated clutches at WB. Estimated sex ratios from artificially incubated clutches at EB (2.5 ± 3.8 % females in 2008 and 23.4 ± 28.3 % females in 2009) were lower than from those artificially incubated clutches at WB (41.7 ± 23.5 % females in 2008, and 57.7 ± 26.6 % females) and from those *in situ* clutches (53.6 ± 28.5 % in 2008, and 72.9 ± 30.7 % females). However, since no clutches were incubated *in situ* in EB we could not ascertain whether there were significant differences or not on likely sex ratios between artificially incubated versus *in situ* incubated clutches. Clutch relocation is currently the only viable conservation option for clutches on EB due to intense egg take, but steps are needed to ensure that natural sex ratio is not distorted. However, on the WB of JNP, *in situ* clutch incubation seems possible through beach protection.

Hawksbill nesting population of Saona Island also face egg take as a major threat. Hence, a similar official conservation program, including artificial incubation, was also carried out for hawksbill clutches at Saona. In a four-year period study, a total of 400 hawksbill nests were recorded in 5 sampling areas of nesting beaches at the south coast of Saona. The majority of clutches studied were artificially incubated (n = 163), while 146 clutches had already been predated by humans when found, and the rest were camouflaged and left for incubation *in situ*. However, due to different factors such as egg take or erosion caused by storms, only a total of 49 clutches incubated *in situ* were studied. In total, 12,340 hatchlings were produced under artificial incubations and released to the sea (1731 in 2007, 4522 in 2008, 2664 in 2009 and 3423 in 2010). No significant effects of artificial incubation in hatching success were detected between years or incubation type. Likewise, there was no significant difference in emergence success between years or incubation type compared to *in situ* incubation. These results indicate that handling was correctly carried out and that incubation conditions in boxes were suitable for hawksbill embryos development. On the other hand, in 21 of the 22 artificially incubated clutches mean temperature during the thermosensitive period was lower than the pivotal temperature derived from laboratory studies, suggesting a male bias in artificially incubated clutches. Current level of egg take is unsustainable for the long term preservation of this nesting population. The low percentages of female hatchling production inferred from clutch temperature data calls for urgent changes in the conservation strategy adopted in Saona.

Studies of *in situ* hatching success leaded to determine the reproductive value of the western beaches of JNP for the leatherback turtle as well as allowing obtaining data on beach parameters potentially affecting clutch incubation. A total of 64 leatherback clutches spanning three nesting seasons (2007, 2008 and 2009) were studied. The hatching success on these beaches appeared to be driven mainly by the effects of beach sector (i.e, La Cueva beach vs. Bahía de las Águilas beach), the incubation duration, and the date of lay. Hatching success of clutches incubated in the beach sector of La Cueva was higher than in those incubated in Bahía de las Águilas. In addition to beach sector, hatching success was strongly influenced by incubation duration; longer incubation durations resulted in lower hatching success. Date of lay affected hatching success, with clutches laid earlier in the nesting season having higher hatching success. The results of this study show that clutches of leatherback turtles in the JNP western beaches presented unusually high hatching success (75.2%) for this species, compared to other rookeries in the Caribbean and elsewhere. This study is particularly relevant in relation to La Cueva beach; this sector hosts 20% of the total clutches laid at western beaches of the Park and demonstrated the highest hatching success levels. However, it is less protected because it is located in the buffer zone out of the Park limits and deserves more protection. Given the exceptional value of hatching success and the current and potential threats affecting leatherback nesting beaches, additional efforts in regulation and management of the protected area are needed.

Understanding spatial and temporal habitat-use patterns to protect both foraging and nesting grounds of reproductive individuals is crucial to success in marine turtle conservation. During the conservation project of marine turtles nesting in DR a total of 10 hawksbill females were satellite tagged and tracked after nesting. For the study of the habitat-use patterns we obtained locations from a total of 9 turtles, 8 tagged in Saona Island in August-September 2008 and September-December 2009, and 1 in JNP in September 2009. Minimum Convex polygon (MCP) analysis showed that individual internesting areas occupied by the turtles ranged from 51.0 to 644.0 km^2 (mean ± SD: 254.3 ± 173.6). Nesting hawksbills of Saona Island remained in the adjacent waters to their nesting beaches using small home range areas during internesting intervals. Core activity areas occurred in shallow waters mainly within 200 m isobaths and associated with coral reefs at the eastern-most tip of the island. The home ranges overlapped, showing similar location and extension both across turtles and years. The results obtained support previously described internesting behaviour for the hawksbill turtle in other internesting areas in the Caribbean. Overall, the common core-use area during internesting was situated inside the DENP's maritime boundaries.

Hawksbill turtles tagged in Saona showed a within-rookery dichotomy in their migratory strategies, with some turtles remaining close to nesting sites in waters of the DR and others migrating to international foraging grounds. Most of the turtles that migrated internationally foraged in waters off Nicaragua and Honduras (n = 5). With the exception of two turtles with 2.2% and 91.5% of the tracked days inside a protected area (Seaflower Biosphere Reserve and Miskitos Cays respectively), the rest of the turtles were in non-protected waters during their entire foraging period. The two turtles that remained in waters of the DR after nesting in Saona stayed within coastal reef ecosystems. One of these turtles stayed inside the JNP while the foraging area of the second one was located in waters adjacent to Bahía de las Calderas (south DR coast) outside any marine protected area. The remaining tracked turtle travelled northwest to the Bahamas where its core-use area was also out of any protected area. This information highlights the relevance of DR protected areas for the conservation of hawksbill‘s internesting and foraging habitats, showing the need of enforcing existing legislation for the protected areas in the country. The present study also corroborates that the waters off Nicaragua and Honduras are exceptionally important foraging areas for the species in the Caribbean, as well as showing the turtles' vulnerability in these waters.

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RESUMEN

INTRODUCCIÓN

Es un hecho irrefutable que las entidades que integran la biosfera se encuentran bajo una gran amenaza de disminución y, en muchos casos, de desaparición definitiva (Larsen et al. 2012; UICN 2012). La pérdida de biodiversidad está siendo particularmente importante en mares y océanos, los cuales albergan la mayor diversidad de ecosistemas y especies del planeta (Agardy 2005). Paradójicamente, el hecho de que los océanos sean patrimonio de todas las naciones crea un vacío legal que se erige como principal obstáculo para incrementar la protección de los mismos y de las especies que viven en ellos (Hendriks et al. 2006; Norse 2010).

Una herramienta indispensable para la conservación de los ecosistemas marinos es la creación y manejo de Áreas Marinas Protegidas (AMPs), que se definen como ''áreas de los océanos designadas para mejorar la conservación de los recursos marinos'' (Lubchenco et al. 2003). En el ámbito concreto de la conservación de los vertebrados marinos, las AMPs se han convertido en una herramienta fundamental para su protección (Abdulla et al. 2008; Botsford et al. 2009). Es evidente que no se puede proteger algo si se desconoce su distribución, por lo que la conservación de especies marinas de gran movilidad exige un exhaustivo estudio previo de los patrones de uso espacial de su hábitat (Costello et al. 2010; Block et al. 2011), especialmente cuando el objetivo es proteger especies migratorias. Este es el caso de las tortugas marinas.

Las tortugas marinas son reptiles adaptados a la vida marina que habitan la Tierra desde el Cretácico (Hirayama 1998). En la actualidad existen siete especies agrupadas en dos familias: familia Dermochelydae, con la tortuga laúd (*Dermochelys coriacea*) como única especie viviente, y la familia Cheloniidae, con seis especies actuales: tortuga carey (*Eretmochelys imbricata*), tortuga bastarda (*Lepidochelys kempii*), tortuga golfina (*Lepidochelys olivacea*), tortuga boba (*Caretta caretta*), tortuga verde (*Chelonia mydas*) y tortuga plana (*Natator depressus*) (Pritchard 1996; Spotila 2004). Las tortugas marinas dependen tanto de hábitats marinos como terrestres para su crecimiento y desarrollo. Estos hábitats incluyen playas, arrecifes bentónicos y mar abierto. El ciclo de vida es similar en las siete especies, con pequeñas variaciones en la duración de las fases, presentando todas ellas un comportamiento de anidación muy estereotipado (Miller 1997; Meylan y Meylan 1999; Ver figura 1.1 en página 51).

Las tortugas marinas son reconocidas como "especies clave" debido al impacto ecológico que tienen en el mantenimiento de la estructura y funcionamiento de los ecosistemas marinos (Bjorndal y Bolten 2003). En las últimas décadas, estas funciones se están viendo alteradas debido a la drástica disminución de muchas poblaciones de tortugas en todo el mundo (McClenachan et al. 2006). La disminución de estas especies se debe a múltiples factores, que van desde la sobreexplotación humana (consumo durante siglos de sus huevos, carne y otros productos derivados de estos animales), hasta la captura accidental por barcos de pesca, el desarrollo costero, la contaminación o los efectos del cambio climático (Wallace et al. 2011).

La temperatura es un factor ambiental crucial en el desarrollo de las tortugas marinas, ya que estas exhiben lo que se denomina determinación sexual dependiente de la temperatura (TSD, por sus siglas en inglés), es decir, no existe una determinación genotípica del sexo, sino que la proporción de sexos primaria está influenciada por la temperatura experimentada por los huevos durante, aproximadamente, el tercio medio del periodo de incubación (Bull 1980; Yntema y Mrosovsky 1982). La temperatura pivote es aquella en la que el 50% de los neonatos que nacen son hembras y el 50% machos. Esta temperatura se sitúa en torno a los 29°C para todas las especies; las temperaturas de incubación inferiores a la temperatura pivote producen más machos y las temperaturas superiores producen más hembras (Yntema y Mrosovsky 1982). Considerando este rasgo de las tortugas marinas, cualquier factor causante de variaciones en la temperatura de incubación afectaría a la proporción de sexos de los neonatos.

Alrededor del mundo se han establecido innumerables programas de conservación con el objetivo de proteger las poblaciones nidificantes de tortugas marinas amenazadas. Una de las prácticas de protección más extendidas ha sido el traslado y reubicación de las nidadas amenazadas a corrales cerrados y protegidos, así como la incubación de los huevos en cajas de poliestireno (Brown et al. 2011; Maulany 2012), lo que puede tener resultados positivos en la conservación. Sin embargo, el traslado de las nidadas puede afectar negativamente al desarrollo embrionario. No solo es posible que se produzca un sesgo de los neonatos hacia uno u otro sexo debido a variaciones en la temperatura, como hemos descrito anteriormente, sino que, además, tanto estas variaciones de temperatura como el manejo y traslado de los huevos pueden afectar negativamente al desarrollo embrionario, al éxito de eclosión y al éxito de emergencia (Eckert y Eckert 1990; Rees y Margaritoulis 2004; Pintus et al. 2009; Santidrián Tomillo et al. 2009). A pesar de estos aspectos negativos, la falta de recursos para la adecuada conservación de las nidadas *in situ* hace que muchos proyectos de conservación, particularmente los establecidos en países en vías de desarrollo, no tengan otra opción que la de incubar los nidos artificialmente, bien en corrales ubicados en las mismas playas o en cajas (García et al. 2003; Brown et al. 2012). En aquellos casos en los que la incubación artificial sea la única estrategia de protección posible es esencial estudiar sus efectos sobre el éxito de eclosión y las proporciones de sexo de los neonatos.

En la región del Caribe habitan seis especies de tortugas marinas: la tortuga boba, la tortuga verde, la tortuga carey, la tortuga bastarda, la tortuga golfina y la tortuga laúd. Históricamente, la presencia de grandes colonias de anidación a lo largo de las costas caribeñas ha sido importante y, sin embargo, hoy en día, pocas poblaciones presentan una gran abundancia de tortugas nidificantes (McClenachan et al. 2006). Si tenemos en cuenta todas las especies, aproximadamente la mitad del total de playas de anidación conocidas presentan menos de 25 rastros por año, lo que equivale a menos de 10 hembras reproductivamente activas (Dow et al. 2007). En el Caribe las tortugas marinas han sido objeto de explotación durante muchos siglos, lo que ha dado lugar a una reducción a gran escala en el número de individuos de sus poblaciones (Parsons 1962; Eckert 1995; Jackson

1997; Meylan 1999; Bell et al. 2006). Las tortugas han sido muy valoradas tanto por sus huevos, como por su carne y caparazón (Fleming 2001; Bräutigam y Eckert 2006). A pesar de encontrarse estrictamente protegidas mundialmente (todas estas especies se enumeran en el Apéndice I de CITES, que prohíbe su comercio a nivel internacional), en algunos países del Caribe todavía se mantiene el comercio de productos de tortuga carey (Bräutigham y Eckert 2006). A estas amenazas directas se le suman los excepcionalmente altos niveles de desarrollo turístico que caracterizan esta región y que han provocado una degradación ambiental irreversible afectando a los hábitats de nidificación de las tortugas marinas (Davenport y Davenport 2006; Bell et al. 2007; Mathenge et al. 2012).

Recientemente se ha establecido que en el suroeste occidental del océano Atlántico la tortuga laúd presenta un estado de preocupación menor para su conservación (IUCN 2013). En la región del Caribe se considera que las poblaciones de esta especie están controladas, y muchas de ellas en crecimiento (Dutton et al. 2005; McGowan et al. 2008; Wallace et al. 2011). Las poblaciones de tortuga carey y tortuga verde se han descrito como poblaciones robustas, pero bajo amenazas que, si no son eliminadas, podrían tener impactos negativos sobre el estado actual de las mismas a corto plazo (Wallace et al. 2011). A pesar de este aparente buen estado de las poblaciones de tortugas marinas en la región del Caribe, aún existen muchos vacíos de información sobre el estado de conservación de pequeñas poblaciones dispersas, algunas de ellas muy diezmadas y amenazadas (Lagueux y Campbell 2005; Bräutigam y Eckert 2007). La desaparición de estas poblaciones podría tener un importante impacto a nivel regional, por lo que su estudio debería considerarse un objetivo prioritario (Dow y Eckert 2011).

La República Dominicana (RD) está situada en el extremo oriental de la isla de La Española. Es la segunda isla en tamaño dentro de las Antillas Mayores y uno de los puntos calientes de biodiversidad más importantes de América Central. En las zonas costeras de RD se ha descrito la anidación de cuatro especies de tortugas marinas: tortuga laúd, tortuga carey, tortuga verde y tortuga boba (Ottenwalder 1981). El país cuenta con un largo historial de consumo y comercio de estas especies, lo que ha constituido un importante recurso para las comunidades costeras (Ottenwalder 1981; Fleming 2001; Reuter y Allan 2006). La industria de artesanía de concha de tortuga carey ha representado una grave amenaza para la supervivencia de esta especie, documentándose durante décadas la venta de artículos de carey a turistas (Stam y Stam 1992; Domínguez y Villalba 1994; Feliz et al. 2008). De igual manera, ha sido generalizado el consumo de la carne, los huevos y la grasa de las tortugas verde y laúd. La pérdida de hábitat debido al desarrollo costero también constituye otra importante amenaza para los hábitats de anidación de las tortugas marinas en el país (Ottenwalder 1996; Geraldes 2003).

Las tortugas marinas están protegidas legalmente y su comercio está prohibido en el país por la Ley de Pesca de 2005 CODOPESCA (Consejo Dominicano de Pesca y Agricultura). Sin embargo, pese a la existencia de estas leyes y de áreas protegidas, la debilidad institucional se hace patente en el frecuente incumplimiento de la ley de medio ambiente y recursos naturales. Un ejemplo de ello es la enorme expansión del turismo en la costa, que ha llevado a una apropiación de parcelas de terreno protegido en zonas costeras (Heredia 2003).

Los últimos datos sobre el estado de conservación de las poblaciones nidificantes de tortugas marinas en RD son los trabajos llevados a cabo por Ottenwalder (1981, 1983). Los resultados de estos trabajos mostraban un declive de las poblaciones nidificantes de tortugas en el país. Durante tres décadas no se ha realizado ningún otro estudio evaluando el estado de estas poblaciones, aunque han sido constantes las denuncias sobre los elevados niveles de explotación y comercio de productos procedentes de las mismas (Stam y Stam 1992; Domínguez y Villalba 1994). La falta de información actualizada sobre el estado de las poblaciones de tortugas marinas de RD ha sido resaltada de forma reiterada en diferentes evaluaciones sobre la situación de las tortugas marinas en la región del Caribe (McClenachan et al. 2006; Dow et al. 2007). La actualización del estado de conservación de las tortugas marinas en el país es necesaria no solo por la supervivencia de las tortugas, sino por el efecto que podría tener sobre la conservación de estas especies a nivel regional.

JUSTIFICACIÓN Y OBJETIVOS

Esta Tesis Doctoral tiene tres objetivos generales: 1) Llevar a cabo el primer estudio sistemático sobre el estado de conservación y amenazas actuales en las playas de anidación de tortugas marinas en las costas de RD en el Caribe Norte; 2) investigar la biología reproductiva de las especies nidificantes en las playas del Parque Nacional Jaragua (PNJ) y en las playas de isla Saona (Parque Nacional Del Este), y al mismo tiempo evaluar los proyectos de incubación artificial establecidos durante el estudio en estas dos áreas para la protección de los nidos amenazados; 3) estudiar el comportamiento durante el periodo entre puestas de las hembras nidificantes y analizar el uso de sus áreas de anidación y alimentación en relación a las áreas protegidas de la región del Caribe .

Estos objetivos generales se concretan en los siguientes objetivos específicos:

- Identificar las principales colonias nidificantes de tortugas marinas en RD y describir los patrones espacio-temporales de anidación en la actualidad, así como evaluar el posible impacto de las amenazas actuales sobre estas poblaciones nidificantes comparándolo con los estudios previos en el país.
- Evaluar el programa de incubación artificial de nidos de tortuga laúd establecido en las playas del PNJ a través del estudio de los potenciales efectos de esta incubación artificial sobre el éxito de eclosión de los nidos y la proporción de sexos de los neonatos.
- Determinar el éxito reproductivo de *Eretmochelys imbricata* en la isla Saona a través del estudio del éxito de los nidos. Evaluar el programa de incubación artificial establecido de la misma manera que para la tortuga laúd en el PNJ.
- Estudiar los factores que afectan al éxito de eclosión de los nidos de tortuga laúd incubados en las playas del oeste del PNJ

con el objetivo de determinar el valor reproductivo de estas playas para la especie. Destacar el papel de la tortuga laúd en las playas amenazadas del PNJ como un vehículo para la conservación de los ecosistemas y la gestión en esta área protegida.

- Investigar el comportamiento y las preferencias de hábitat de las hembras de tortuga carey nidificantes de isla Saona durante los periodos entre puestas. Establecer las áreas preferentes de uso con respecto a los límites de las áreas protegidas en sus áreas de anidación y de alimentación.

MATERIALES Y MÉTODOS

Área de estudio

República Dominicana comparte con Haití la isla de La Española, que forma parte del archipiélago de las Antillas Mayores en la región del Caribe. El país tiene una extensión de 48.445 km^2 y una población estimada de unos 10 millones de personas. Cuenta con 1.576 km de costa. Tiene un clima tropical con una fuerte influencia marítima, una temperatura promedio de 25 °C y rangos de precipitación que oscilan entre los 350 mm a los 2.500 mm. El Sistema Nacional de Áreas Protegidas cuenta con 86 unidades englobadas en seis categorías con diferentes niveles de protección (desde áreas de protección estricta hasta paisajes protegidos).

El presente estudio se llevó a cabo principalmente en las áreas protegidas del PNJ e isla Saona. El PNJ se encuentra situado en el suroeste de RD, comprende 1.374 km^2, de los cuales 905 km^2 son marinos. El parque abarca numerosos ecosistemas que destacan por el elevado número de especies endémicas presentes; estos ecosistemas incluyen bosques naturales, playas, costas rocosas, humedales, pastos marinos y arrecifes de coral. El parque cuenta con dos áreas de playas, el área de Bahía de las Águilas en el extremo occidental y las playas próximas a la laguna de Oviedo al este; ambas proporcionan unas condiciones favorables para la anidación de las tortugas marinas laúd, carey y verde. El Parque Nacional del Este (PNDE, sureste de RD), se fundó el 16 de septiembre de 1975. En este parque se incluye la isla Saona que, con una superficie de 110 km^2, es la mayor de las islas pertenecientes a RD. Las zonas neríticas adyacentes a las playas de la isla comprenden arrecifes de coral y praderas de pastos marinos que proporcionan hábitats propicios para cientos de especies de plantas, aves, peces y otros animales marinos. Isla Saona es también el hogar de varias especies animales y vegetales endémicas, amenazadas o en peligro de extinción. Las playas de la isla son un hábitat favorable para la anidación de la tortuga carey y verde.

Muestreos de playas

Antes de iniciar los muestreos de playa se revisó la información de los estudios llevados a cabo previamente por Ottenwalder (1981), y se realizaron entrevistas a guardaparques y pescadores de las áreas de anidación descritas en tales estudios. Se realizaron recorridos a pie durante el día en las playas del PNJ y Saona durante cinco temporadas de anidación consecutivas, 2006-2010. En las playas del este del PNJ, en las temporadas de 2006, 2007 y 2008 los recorridos se realizaron una vez por semana. A partir de 2009, los guarda parques recorrieron diariamente esta playas. Las playas del oeste del PNJ se recorrieron entre 3 y 4 veces por semana. En la isla Saona, las playas se recorrieron al menos una vez por semana durante todo el año; a partir de 2008 estos recorridos se incrementaron a 3-4 por semana durante el periodo de máxima anidación (de junio a noviembre).

Durante estos recorridos se registró información de todos los rastros de tortugas encontrados, identificándose cada especie mediante el estudio de características propias del rastro de las mismas (Pritchard y Mortimer 1999). Para determinar el número de nidos en playa se diferenció entre rastros de tortugas con anidación exitosa (depositaron huevos) de aquellos rastros de tortuga sin anidación exitosa. Debido a las limitaciones de financiación sólo fue posible llevar a cabo recorridos nocturnos en isla Saona y en las playas occidentales del PNJ.

La verificación de presencia de anidación en otras áreas de RD se llevó a cabo mediante visitas, recorridos esporádicos y entrevistas a pescadores y habitantes de las comunidades locales durante el periodo 2006-2010. Este trabajo se realizó en 11 zonas de la costa norte en 2006, 2007 y 2008, y en seis playas de las costas este y sur del país en 2008, 2009 y 2010. Los informes sobre otros eventos de anidación, en particular aquellos procedentes de hoteles o complejos turísticos situados frente a la playa, también fueron registrados.

Toma de datos y traslado de nidadas

Los nidos encontrados se posicionaron mediante coordenadas GPS. Se recogieron datos sobre la fecha y la hora de puesta, ubicación del nido en relación a la línea de marea alta (distancia), ubicación a lo largo de la playa (sector) y zona (vegetación, borde, arena). Se utilizó una cinta métrica flexible para tomar las medidas en curvo de cada tortuga: longitud curva de caparazón (LCC) y ancho curvo de caparazón máximo (ACCmax). Las tortugas laúd se marcaron con placas de metal Inconel grandes (Marca nacional y Tag Co., Newport, EE.UU.) entre la cola y aletas traseras. Las tortugas carey se marcaron con marcas de metal Inconel pequeñas en el borde de las dos aletas delanteras.

Una vez que la tortuga abandonó la playa se procedió al manejo de los huevos. El destino de las nidadas se decidió en función del riesgo de inundación del nido por la marea, de la presencia de depredadores y/o de la distancia a campamentos pesqueros y poblaciones. Si no se detectaron amenazas para los huevos, la nidada se dejó en la playa para su incubación en condiciones naturales (*in situ*), camuflando el rastro de la tortuga para evitar su detección. En aquellos casos en los que existía riesgo de pérdida de los huevos por alguna amenaza, se procedió a la excavación del nido para el traslado de los huevos a las casetas de los guarda parques, donde se incubaron en cajas de poliuretano (dimensiones: 30 cm ancho x 50 cm largo x 32 cm profundidad). El manejo y traslado de huevos se hizo con extremo cuidado, siguiendo la metodología descrita, para minimizar la mortalidad de embriones (Abella et al. 2007). Los laterales y parte inferior de las cajas se cubrieron con arena procedente del nido original para evitar el contacto de los huevos con las paredes. Igualmente, una vez en la caja, los huevos se cubrieron también con arena del nido original. Cada caja fue etiquetada con un código, indicando el nombre de la playa, la fecha de puesta y el número de huevos incubados. Las cajas fueron revisadas diariamente a lo largo de todo el periodo de incubación y las tapas se abrieron durante dos o tres horas al día para permitir la circulación del aire.

Éxito de eclosión y éxito de emergencia

En los nidos incubados artificialmente, se procedió al estudio de los huevos siempre 48 horas después de la emergencia del último neonato. Las crías fueron liberadas cuando entraron en estado de frenesí y, siempre que fue posible, las liberaciones se realizaron en las playas de procedencia de los nidos. Antes de su liberación, se escogieron aleatoriamente 20 neonatos y se tomaron medidas de la longitud recta del caparazón (LRC) con un pie de rey con una precisión de 0,1 cm. Estos 20 neonatos se pesaron con una balanza electrónica con una precisión de 0,1 g. Los nidos que se dejaron incubando *in situ* se controlaron durante los muestreos de playa; cuando se observaron rastros de neonatos saliendo del nido se llevó a cabo la excavación y estudio del contenido. La clasificación del contenido de los nidos y el cálculo del éxito de eclosión y éxito de emergencia se detallan en las páginas 81, 82 y 83 del capítulo 3.

Estimación de la proporción de sexos de los neonatos

Debido al estado de conservación de la población nidificante de Saona no se consideró la opción de sacrificar neonatos para la determinación del sexo mediante histología. La determinación de la proporción de sexos se hizo mediante métodos indirectos, utilizando el periodo de incubación (PI). Para cada nido, se calculó el PI (definido como el número de días entre la puesta de huevos y la primera eclosión de neonatos). Dado que no existe ningún estudio previo sobre el PI pivote (PI en el que nacen 50% machos y 50% hembras) para Saona, utilizamos el valor estimado para el área de anidación de carey más próxima, Isla de Mona (Mrosovsky et al. 2009), y en el caso de la tortuga laúd se utilizó la curva que relaciona PI y proporción de sexos estimada para la población de Suriname (Godfrey 1997). Se extrapolaron sobre la curva los valores de PI registrados para cada nido, obteniendo de esta manera la proporción de sexos correspondiente a cada PI.

Telemetría por satélite

Se colocaron transmisores vía satélite a tortugas carey interceptadas en el momento de desove en las playas de isla Saona, tras haber realizado las puestas, siguiendo la metodología de Godley et al. (2003).

Las posiciones de cada tortuga se determinaron a través del sistema ARGOS. Este sistema de localización y recolección de datos por satélite asigna un valor de precisión a las posiciones estimadas (*location class* [LC] 3, 2, 1, 0, A, B, Z). Los datos de teledetección se descargaron y se filtraron mediante la herramienta de análisis del seguimiento por satélite (STAT) (Coyne y Godley 2005). En este estudio las LCs seleccionadas para la estimación de las áreas de uso fueron 3, 2, 1, A y B (señaladas en trabajos previos como las más precisas en estudios de carey; Gaos et al. 2012).

Para estimar el área de actividad de las hembras nidificantes dentro de las zonas entre puestas y las zonas de alimentación se utilizó la herramienta de mínimo polígono convexo (MPC). La intensidad de uso de las áreas de actividad se estableció mediante el estimador de densidad kernel (EDK) (Worton 1989). Las distribuciones de densidad fueron representadas en los mapas utilizando los contornos de distribución del 50% y 90% (Hooge et al. 2000).

La ubicación de las tortugas con respecto a las áreas marinas protegidas (AMP), se analizó superponiendo los límites de estas áreas en todos los mapas resultantes. Los límites de las AMPs fueron descargados de la base de datos mundial de áreas protegidas (www.wdpa.org). Para llevar a cabo estos análisis se utilizó el programa de sistemas de información geográfica ArcGIS v.10.0 (ESRI 2010).

RESULTADOS Y DISCUSIÓN

Los resultados de este estudio confirman la presencia de tres especies de tortugas marinas anidando actualmente en RD: la tortuga laúd, la tortuga carey y la tortuga verde. El PNJ presentó el mayor número de nidos por año de tortuga laúd (media = 126,4 ± 74,1, rango: 17-210) con un total de 632 nidos contabilizados durante el periodo 2006-2010. El número total de nidos de tortuga carey registrados en el mismo periodo en el PNJ fue de 73 (media 14,6 ± 6,7, rango: 7-22). En los cinco años de estudio solo se registró un nido de tortuga verde en esta área. En las playas de isla Saona se registró el mayor número de nidos de tortuga carey (media 100 ± 8,4, rango: 93-111, n = 400) y tortuga verde (media = 9,2 ± 6,2, rango: 1-15, n = 37) de todo el país, mientras que de tortuga laúd se registraron 22 nidos en el mismo periodo (media = 5,5 ± 4,8, rango: 1-11). La anidación de las diferentes especies registrada en el resto de playas del país es muy reducida. El número de nidos por playa se presenta en la figura 4.2 de la página 96. El periodo de anidación de la tortuga laúd en el PNJ se extiende desde marzo hasta agosto con la mayoría de nidos registrada entre los meses de abril y junio (89,6% de los nidos). En la isla Saona, la tortuga carey fue observada anidando a lo largo de todo el año con la mayoría de nidos registrada en los meses comprendidos entre junio y noviembre. La tortuga verde anida en Saona de julio a noviembre, con el máximo de nidos registrados en el mes de agosto. Tanto en el PNJ como en Saona, todas las tortugas detectadas en los muestreos realizados durante el periodo de estudio fueron identificadas con una marca metálica numerada. La media de LCC de la tortuga laúd en el PNJ es de 147,4 ± 8,7 cm, n = 13. En Saona se midió el LCC de un total de 19 hembras de carey (media 87,2 ± 4,7 cm). La talla media de las tortugas carey que anidan en RD es similar a la registrada en otras áreas de anidación del Caribe. En el caso de la tortuga laúd, la talla media registrada en el PNJ es inferior a la registrada en la mayoría de las poblaciones del mundo. Este dato podría interpretarse como presencia de una elevada proporción de hembras neófitas, lo que indicaría una posible recuperación de la población.

La depredación de huevos por humanos se identificó como la principal amenaza sobre los nidos de tortugas marinas en playas de anidación de RD. En el PNJ, los nidos de las playas situadas en el extremo oriental vienen sufriendo los niveles más altos de depredación. El número total de nidos depredados en playa por año de cada una de las tres especies estudiadas se muestra en la tabla 4.2 de la página 99. Tanto en el PNJ como en Saona se encontraron restos de adultos de tortuga carey y tortuga verde presumiblemente depredados para consumo humano.

Los resultados de este estudio muestran que la anidación de tortugas marinas en el país está restringida a las áreas protegidas y poco desarrolladas del PNJ e isla Saona. Cuando comparamos los resultados obtenidos en este trabajo con los obtenidos en trabajos previos llevados a cabo en el país, se observa un declive de las poblaciones nidificantes de tortugas marinas. Durante el periodo de estudio se observó un manejo inadecuado de las áreas protegidas por parte de las autoridades locales, registrándose un alto grado de depredación y consumo tanto de huevos como de carne de tortugas adultas por habitantes locales. Estos resultados ponen de relieve la necesidad de una mejor protección y manejo de estas áreas.

Como medida frente a la elevada depredación humana se continuó un programa de protección de nidos establecido desde 1974, en el que los huevos eran trasladados desde la playa para su incubación en cajas. Este sistema de protección de nidos de tortuga laúd se ha realizado voluntariamente durante treinta y ocho años de forma esporádica bajo escaso control científico. En el presente estudio hemos tratado de evaluar la eficacia de esta medida de conservación investigando si la incubación artificial puede alterar el éxito de eclosión y las proporciones de sexo resultantes en comparación con los nidos incubados en playas.

Durante el periodo de estudio se estudiaron un total de 109 nidos de tortuga laúd (puestos en las temporadas 2008 y 2009): 35 incubados artificialmente en las playas del este (PE), 31 incubados artificialmente en las playas occidentales (PO) y 43 incubados *in situ* también en las playas occidentales del parque. Los resultados obtenidos muestran que el método de incubación influyó significativamente en el éxito de eclosión de las nidadas (GLMM: $\chi^2 = 76{,}7$, $p < 0{,}001$). El éxito de eclosión fue significativamente más alto en los nidos incubados *in situ* que en los incubados artificialmente, tanto en PO como en PE. El método de incubación tuvo también un efecto sobre el número de embriones muertos en estado de desarrollo temprano (GLMM: $\chi^2 = 16{,}1$, $p < 0{,}05$). Los nidos incubados artificialmente en PO tenían más embriones muertos en estado temprano que los nidos incubados *in situ* y los incubados artificialmente en PE. El número de embriones muertos en estado tardío también se vio afectado por el método de incubación (GLMM: $\chi^2 = 67{,}4$, $p < 0{,}001$). Los nidos incubados artificialmente en PE y PO presentaron un número significativamente mayor de embriones muertos en estado tardío que los nidos incubados *in situ*. Igualmente, el método de incubación afectó a la duración de la misma (GLMM: $\chi^2 = 16{,}0$, $p < 0{,}01$). Las temperaturas de incubación fueron significativamente inferiores en los nidos incubados artificialmente en PE en comparación con los incubados artificialmente en PO (prueba t de Welch, $t_{22,9} = -5{,}66$, $p < 0{,}001$). El lugar de incubación de los nidos incubados artificialmente en PO está situado en una zona elevada de matorral árido, por lo que los nidos alcanzaron altas temperaturas durante la incubación. Como resultado, la duración de la incubación y las proporciones de sexo de los neonatos incubados en esta ubicación fueron similares a las obtenidas en los nidos incubados *in situ*. Los nidos incubados artificialmente en PE tuvieron duraciones de incubación significativamente más largas que los nidos incubados artificialmente en PO y los incubados *in situ*. Esto se tradujo en porcentajes de hembras muy bajos en PE, probablemente debido a que el lugar de incubación estaba ubicado en una zona de mayor humedad (bosque junto a la laguna de Oviedo). Dado que en estas playas no se incubaron nidos *in situ*, no se pudo comprobar las diferencias en la proporción de sexos entre nidos incubados en condiciones naturales y nidos incubados artificialmente.

En las playas occidentales del PNJ la incubación de los nidos *in situ* parece posible mejorando sensiblemente la protección de playas. Sin embargo, en las playas del este del parque esta opción no es viable actualmente debido a la intensa depredación humana que existe. En las playas orientales del PNJ la incubación artificial parece la única opción para la incubación con éxito de los huevos, aunque, teniendo en cuenta los bajos porcentajes de hembras que se obtienen en los nidos incubados artificialmente, es necesario implantar medidas de conservación que aseguren que la proporción de sexos de los neonatos sea similar a la producida en los nidos incubados *in situ*.

Al igual que ocurre en las playas del PNJ, la colonia reproductora de tortuga carey de isla Saona se encuentra seriamente amenazada principalmente por la depredación humana de huevos. En 2007 se inició en Saona un programa de incubación de huevos en cajas como medida urgente para proteger los nidos e incrementar el reclutamiento de neonatos al mar.

Durante los cuatro años de estudio se registraron 400 nidos de tortuga carey. En total se incubaron artificialmente 163 nidos, otros 146 se encontraron ya depredados por humanos y 91 se dejaron incubando *in situ*. La figura 6.3 de la página 140 muestra la variación anual en estos porcentajes. En todos los nidos depredados por humanos se depredó el 100% de los huevos.

En estos cuatro años de estudio se liberaron al mar un total de 12340 neonatos. Las biometrías de los neonatos liberados sugieren que éstos son más pequeños que los de otras áreas de puesta del Caribe (2008: LCC = 3,8 ± 0,1 cm, rango: 3,2-4,2, n = 480); 2009: 3,8 ± 0,2 cm, rango: 3,0-4,0, rango: 12,6-18,2, n = 160). No se detectaron efectos significativos de la incubación artificial sobre el éxito de eclosión entre años (ANOVA: $F_{(3,157)}$= 1,45, P = 0,383), tipo de incubación ($F_{(1,157)}$ = 0,7, P = 0,45); o su interacción ($F_{(3,157)}$ = 0,52, P = 0,669). Igualmente no hubo diferencias significativas en el éxito de emergencia entre años (ANOVA: $F_{(3,157)}$ = 1,1, P = 0,469), el tipo de incubación ($F_{(1,157)}$ = 1,28, P = 0,327); o la interacción de ambos factores ($F_{(3,157)}$ = 0,68, P = 0,563), comparado con los nidos incubados *in situ*. Se obtuvo el periodo de incubación (PI) de un total de 84 de los nidos incubados artificialmente y de 24 de los nidos incubados *in situ*. Se observaron periodos de incubación más largos en los nidos incubados en cajas que en los incubados *in situ*. Por otro lado, en 21 de los 22 nidos en los que se registró la temperatura de incubación durante el periodo termosensible, ésta fue menor que la temperatura pivote estimada en estudios realizados en el laboratorio recogidos en la literatura (Mrosovsky et al. 2009). Los resultados de periodos y temperatura de incubación sugieren un sesgo hacia la producción de machos en estos nidos incubados artificialmente.

Los actuales niveles de depredación de huevos por humanos no son sostenibles para la conservación a largo plazo de esta colonia reproductora. Sin embargo, los bajos porcentajes de hembras obtenidos en los nidos incubados en cajas muestran la necesidad de cambios urgentes en la estrategia de conservación adoptada.

En los últimos treinta años, RD ha sufrido un desmesurado desarrollo de lo que se conoce como “turismo de masas”, especialmente en las costas norte y este del país. Recientemente, este modelo de turismo, junto con la explotación de minas de bauxita, ha sido propuesto como vía para el desarrollo de la región suroeste del país. Dentro de estos planes de desarrollo se incluyen proyectos para la construcción de complejos turísticos dentro del área protegida del PNJ. En el presente estudio se analizó el éxito de eclosión de 64 nidos de tortuga laúd puestos durante tres temporadas de anidación (2007, 2008 y 2009). Los resultados muestran que el éxito de eclosión de los nidos estudiados parece estar determinado por tres factores: el sector de la playa donde se incubaron, la duración del PI y la fecha de puesta. El éxito de eclosión de los nidos incubados en el sector de La Cueva fue significativamente más alto que el de los nidos incubados en el sector de Bahía de las Águilas. El PI afectó al éxito de eclosión, los nidos con PI más largos presentaron éxitos de eclosión más bajos. La fecha de puesta también afectó al éxito de eclosión: los nidos depositados al comienzo de la temporada de puesta presentaron éxitos de eclosión significativamente superiores a los incubados al final de la temporada. Los nidos de tortuga laúd incubados en las playas occidentales del PNJ presentan un éxito de eclosión inusualmente alto (75,2 %) para esta especie. Los resultados de este estudio son especialmente relevantes en relación a la playa de La Cueva (figura 7.1 en página 158). Este sector recibe el 20% del total de nidos de tortuga laúd en las playas occidentales del PNJ, nidos que además registraron los niveles más altos de éxito de eclosión. Sin embargo, este sector está menos protegido, ya que se encuentra en la zona de amortiguamiento del PNJ, fuera de los límites del área protegida. Teniendo en cuenta los excepcionales niveles de éxito de eclosión y las amenazas actuales y potenciales que afectan a las playas de anidación de tortuga laúd, se requiere un esfuerzo adicional en la regulación, gestión y redefinición de esta área protegida.

Realizar una protección eficaz de especies migratorias requiere conocer los patrones espaciales y temporales de uso de hábitat de las mismas en las diferentes etapas de su ciclo vital. De estos estudios depende en gran medida la eficacia en la gestión y conservación de las áreas marinas protegidas (AMPs) (Maxwell et al. 2011; Scott et al. 2012). En este trabajo se estudió el comportamiento entre puestas y post-puesta de las tortugas carey nidificantes en Saona. Se colocaron transmisores de satélite a un total de 9 hembras nidificantes: 8 en Saona, en 2008 y 2009; y 1 en el PNJ en 2009.

Durante los periodos entre puestas las hembras nidificantes de tortuga carey en Saona permanecieron en las aguas adyacentes a sus playas de anidación. Los MPCs obtenidos muestran que las áreas ocupadas por las tortugas entre puestas variaron entre 51,0 y 644,0 km^2 (media = 254,3 ± 173,6). Las áreas de mayor intensidad de uso de las tortugas nidificantes en Saona se localizaron en el extremo oriental de la isla. Estas áreas se caracterizaron por ser aguas poco profundas, situadas principalmente dentro de la isobata de los 200 m de profundidad, y estar asociadas a arrecifes de coral. Las áreas de distribución se solaparon y su ubicación y extensión fueron similares entre tortugas y entre años. Nuestros resultados confirman el comportamiento de tortugas carey nidificantes durante el periodo entre puestas descrito previamente en otras áreas del Caribe (van Dam et al. 2008; Marcovaldi et al. 2012; Walcott et al. 2012). Dentro de este hábitat entre puestas descrito, los EDKs del 90% y del 50% tuvieron una extensión estimada de 81,7 km^2 y 32,2 km^2 respectivamente. Esta área de uso común se localizó dentro de los límites del área marina del PNDE.

Las tortugas mostraron una dicotomía en sus estrategias migratorias: algunas permanecieron relativamente cerca de sus playas de anidación, en aguas de RD, y otras migraron a zonas de alimentación internacionales. La mayoría de las tortugas migraron a áreas de alimentación situadas en aguas de Nicaragua y Honduras (n = 5). En estas aguas, con la excepción de dos tortugas que estuvieron el 2,2% y el 91,5% de los días dentro de áreas marinas protegidas (Reserva de la Biosfera Seaflower y Cayos Misquitos respectivamente), el resto de tortugas permanecieron durante todo el periodo de alimentación fuera de cualquier área protegida durante sus periodos de alimentación. Una de las tortugas que permaneció durante su periodo de alimentación en aguas dominicanas, se mantuvo dentro de los límites del PNJ, mientras que el área de alimentación de la segunda se situó en aguas adyacentes a Bahía de las Calderas (costa sur de RD), fuera de cualquier área marina protegida. La última tortuga rastreada viajó al noroeste de las Bahamas, donde su área de actividad se mantuvo fuera de áreas protegidas.

La información obtenida pone de relieve la importancia de las áreas protegidas de RD para la tortuga carey, tanto como hábitat entre puestas como hábitat de alimentación, y muestra la necesidad de hacer cumplir la legislación vigente de las áreas protegidas del país. El presente estudio también confirma que las aguas de Nicaragua y Honduras son áreas de alimentación excepcionalmente importantes para la tortuga carey en el Caribe, y hace patente la vulnerabilidad de las tortugas en estas aguas.

CONCLUSIONES

La presente tesis estudia el estado de conservación de las tortugas marinas nidificantes de RD. Las conclusiones del presente estudio son las siguientes:

1. En este estudio se ha realizado la primera evaluación detallada del estado de conservación de las poblaciones de las tortugas marinas nidificantes en RD a partir de muestreos realizados durante el periodo 2006-2010. Se ha constatado la anidación de tres especies de tortugas marinas: tortuga laúd, tortuga carey y tortuga verde; sin embargo, durante el periodo de estudio no se detectó la presencia de tortuga boba, cuya anidación había sido descrita en el país en estudios previos.

2. Actualmente, la anidación de tortugas marinas se concentra en las áreas protegidas del PNJ e isla Saona (PNDE). La anidación en playas fuera de estas áreas es esporádica. Los resultados de este estudio sugieren que se ha producido una gran reducción en la abundancia de las cuatro especies de tortugas marinas en el país desde la década de 1980.

3. El PNJ alberga, con un total de 632 nidos registrados, el mayor número de nidos por año de tortuga laúd de todas las áreas del país estudiadas (media = 126,4 ± 74,1, rango 17-210). El periodo de anidación se extiende desde marzo hasta agosto, registrándose el mayor número de nidos en los meses de abril a junio. El número estimado de hembras de laúd anidando por año varió de 3 a 40. El tamaño de las tortugas laúd en RD (LCC medio = 147,4 cm) es inferior al registrado en la mayoría de las poblaciones del mundo. De este dato podría interpretarse la presencia de una elevada proporción de hembras neófitas, lo que indicaría una posible recuperación de la población. En el PNJ también se registró anidación esporádica de tortugas carey y verde.

4. La isla Saona acoge las playas más importantes del país para la anidación de tortuga carey (media anual = 100 ± 8,4 nidos, rango 93-111, n = 400) y tortuga verde (media 9,2 ± 6,2, rango 1-15, n = 37). La tortuga carey anida en la isla durante todo el año, siendo el periodo de máxima anidación el comprendido entre los meses de junio y noviembre. El número estimado de hembras de carey anidando por año varió entre 21 y 25. La talla media de las tortugas carey que anidan en RD (LCC = 87,2 cm) es similar a la registrada en otras áreas de anidación del Caribe.

5. Las principales amenazas para la conservación de estas poblaciones nidificantes de tortugas marinas son el elevado nivel de depredación de huevos para consumo humano y el desarrollo costero. Como medida de protección frente a la depredación humana se establecieron programas de incubación artificial de nidos.

6. En el PNJ la incubación artificial se llevó a cabo por separado en las playas orientales y occidentales del parque. Se compararon el éxito de eclosión y la proporción de sexos de los neonatos entre los nidos incubados artificialmente y los nidos incubados *in situ.* Los resultados mostraron que en los nidos artificiales incubados en el oeste el éxito de eclosión fue muy bajo, mientras que en los incubados artificialmente en el este los periodos de incubación indicaron un sesgo hacia a la producción de machos.

7. La incubación de los nidos fuera de la playa es la única opción de conservación viable en las playas orientales del PNJ, dada la elevada depredación humana (100% de nidos y huevos), pero es necesario evitar la posible desviación de la proporción de sexos natural que ocasiona la incubación artificial. En las playas occidentales la conservación de los nidos *in situ* parece posible a través de la mejora en el manejo y protección de las playas.

8. El programa de incubación artificial llevado a cabo en Saona permitió la liberación de más de 12000 neonatos de tortuga carey. No se encontraron diferencias en el éxito de eclosión y emergencia entre los nidos incubadas *in situ* y los incubados artificialmente. Sin embargo, las bajas temperaturas y largos periodos de incubación registrados en los nidos incubados artificialmente sugieren un sesgo hacia la producción de machos. A pesar de que la incubación artificial es eficaz en cuanto a la producción de neonatos, la escasa producción de hembras pone de manifiesto la necesidad de mejorar esta medida de protección.

9. Se estudiaron los factores que afectan al éxito de eclosión de los nidos de tortuga laúd incubados en las playas occidentales del PNJ. El sector de la playa donde se incubaron, la duración del periodo de incubación, la fecha de puesta y el número de huevos incubados parecen ser los principales factores que determinan el éxito de eclosión.

10. Las nidadas de tortuga laúd de las playas occidentales del PNJ presentaron un inusual elevado éxito de eclosión para esta especie (75,2 %) comparado con otras poblaciones del caribe (~ 50%). Dado el valor excepcional de éxito de eclosión y las amenazas actuales y potenciales que afectan a las playas de anidación de la tortuga laúd, se requiere un esfuerzo adicional en la regulación y gestión de esta área protegida.

11. El sector de La Cueva alberga el 20% del total de nidos depositados en las playas occidentales del PNJ, presentando además los niveles más altos de éxito de eclosión. Este sector se encuentra fuera de protección en la zona de amortiguamiento, fuera de los límites del parque, por lo que se recomienda su inclusión dentro del PNJ.

12. A través del seguimiento mediante telemetría por satélite de tortugas carey nidificantes en Saona, se constató que, durante el periodo entre puestas, estas tortugas se mantuvieron en las aguas territoriales RD, en su mayoría sobre la plataforma continental (<200 m), en zonas caracterizadas por aguas relativamente poco profundas y cercanas a las playas de anidación.

13. El área de uso común estimada para las tortugas carey nidificantes en Saona durante los periodos entre puestas se situó dentro de los límites del PNDE. Las áreas de uso individual se localizaron en el extremo oriental de la isla y mostraron similar ubicación y extensión entre tortugas y años. Se recomienda aumentar los esfuerzos para mitigar la pesca ilegal y para restringir el tráfico de embarcaciones en estas aguas, a fin de proteger a las tortugas en su hábitat entre puestas.

14. Durante el periodo de alimentación, el 78,0% de las posiciones obtenidas se encontraron fuera de cualquier área marina protegida, tanto en aguas de RD como en aguas internacionales de Bahamas, Nicara-

gua y Honduras. Nuestros resultados destacan la importancia de diferentes áreas protegidas de RD como áreas de alimentación para la tortuga carey, mostrando la necesidad de hacer cumplir la legislación existente y de expandir algunas de las áreas protegidas en el país.

15. El presente estudio también corrobora que las aguas de Nicaragua y Honduras son áreas de alimentación de excepcional importancia para las tortugas carey que anidan en la región del Caribe, y muestra la vulnerabilidad de las tortugas en dichas aguas.

16. A pesar de la importancia de las áreas marinas y costeras protegidas, no sólo para las tortugas marinas nidificantes, sino también para la biodiversidad marina en el país, la gestión de estas áreas es limitada debido a la falta de recursos. La recuperación de las poblaciones nidificantes de tortugas en el país depende de una gestión competente de las áreas protegidas, así como de la aplicación efectiva de las leyes que prohíben el consumo y el comercio de cualquier producto de tortuga marina.

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CHAPTER I:
GENERAL INTRODUCTION

1. 1. BIODIVERSITY LOSS AND CONSERVATION

The maintenance of biodiversity (genetic, population, species and ecosystem diversity) is considered to be one of the highest conservation priorities of our time (Brooks et al. 2006; Larsen et al. 2012). If we revise the biodiversity studies conducted in the last decades we can confirm that the entities that constitute our biosphere are individually and collectively highly endangered or outright lost, with many species declining to critically low levels and with significant numbers becoming extinct (IUCN 2011). In the last four decades, there has been a decrease in individual populations of many species and a large number of habitats have suffered the loss of their original conditions (Brooks et al. 2006; Laurance 2007; Butchart et al. 2010). This biodiversity loss has become one of the most pressing crises, which has led to a growing global concern about the status of the biological resources on which human life depends (ONU 2008; CBD 2010). Habitat loss and fragmentation, overexploitation, the impact of invasive alien species, wildlife trade, pollution and climate change have been identified as the main factors causing the loss of species and ecosystems in both terrestrial and marine environments (Jackson et al. 2001; Dulvy et al. 2003; Lotze et al. 2006; Botkin et al. 2007; Stork et al. 2009; Wlodarska-Kowalczuk et al. 2010).

Overexploitation means harvesting species from the wild at rates faster than natural populations can recover. This threat is associated with the extinction of many endangered vertebrates both in land and marine ecosystems (Butman and Carlton 1995; Rosser and Mainka 2002; McClenachan et al. 2006). Overexploitation presents itself in many forms: exhausting a species used for food, clothing, or medicinal therapies (Alves and Rosa 2007). Sometimes organisms are harvested or captured for purposes other than food and many species are traded to be pets, souvenirs, or trophies (Spurgeon 1992; Fleming 2001).

Although one of the main current challenges in conservation is to increase efforts to stop global biodiversity loss (Hoffmann et al. 2010), recent studies show that the rate of this loss does not seem to be slowing (Butchart et al. 2010). Currently, there is no evidence of mass global extinctions; however, evidence does demonstrate high levels of species lost at the local and regional scale, with corresponding negative effects upon their ecosystems (Stork 2010). This is particularly important in marine ecosystems, where human activities are driving many species to their ecological extinction (Jackson et al. 2001).

1. 2. WHAT IS CONSERVATION?

Conservation is defined (in an anthropocentric and utilitarian way) as "*the management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations*" (IUCN 1980). In response to the biodiversity loss, a new scientific discipline called Conservation Biology was consolidated in the eighties of the 20th century (Soulé and Wilcox 1980). The main objectives of this discipline are to evaluate the factors and reasons of the decline of biodiversity, develop tools and useful measures to prevent its loss, and provide answers to specific questions that can be applied to management decisions (Wilson 1992; Hendriks et al. 2006). The design of protected areas, establishment of breeding programs to prevent loss of genetic variability in small populations, or reconcile the conservation needs with needs of local communities are some of the tools used for the preservation of biodiversity (Primack 1993; Robinson 2006; Griffiths and Pavajeau 2008). The challenges of conservation biodiversity lie in the realm of organizational behaviours, human values, policy-making processes, legal structures, communication flows, public education and agency culture.

1. 3. MARINE BIODIVERSITY CONSERVATION

Most of the Earth's surface (70.8%: 362 million km^2) is covered by oceans and seas, which are home to the planet's greatest diversity of ecosystems and species (Agardy 2005). Currently, about two-thirds of the world's population lives within 60 km of the coast, with many people relying directly on resources provided by the oceans. This high population density in coastal regions is threatening the ecological integrity of the marine environment, which is becoming increasingly degraded due to the compounding effects of various human activities (Rogers and Laffoley 2011). Few corners of the world's oceans remain unaffected (Halpern et al. 2007).

Marine ecosystems are undergoing a great loss of diversity due to direct causes such as overexploitation by fisheries, coastal development, increased pollution and habitat destruction, as well as indirect causes such as climate change (Paramo et al. 2009; Cheung et al. 2010; Polidoro et al. 2010). Overfishing is the primary driver of biodiversity loss in marine systems (Hoffmann et al. 2010). Throughout the 20th century, fishing fleets were transformed from small-scale operators to large industrial ships that employ the use of trawlers, netting, and other high capacity tools. The world's current fishing capacity is estimated to be up to 2.5 times more than what is needed to land a sustainable yield (Sumaila et al. 2012). In addition, bycatch (non-target catch) has contributed to the decline of endangered marine species, including marine turtles, cetaceans, sharks, birds, and lots of other animals over short timescales (i.e. decades) (Lewison et al. 2004; Wallace et al. 2013).

In coastal zones several activities pose threats to marine biodiversity, most of which are a direct result of the increase of human population worldwide and demographic trends (Gray 1997). The escalating population density in the coastal areas has intensified pressure on the use of marine and coastal resources provoking habitat degradation, fragmentation and destruction (Nicholls et al. 2007). In addition, the growth of large coastal cities and industrialisation has brought about a rise in pollution levels - another major threat to marine biodiversity. Human activities are mainly to blame for the entry of chemicals, particles, industrial waste, agricultural and residential waste, plastics and other solid materials into the ocean (Derraik 2002; Cole et al. 2011). Habitat loss represents another alarming threat in marine ecosystems. For instance, about one third of the mangroves, seagrass beds and wetlands worldwide have been lost in recent decades as a result of deforestation and urbanisation (Lewis et al. 2011; Penha-Lopes et al. 2011). Coral reefs, the most biodiverse ecosystems in the ocean, are estimated to harbour around one third of all described marine species (Reaka-Kudla 2001). These reefs are highly threatened and their loss would mean the extinction of much of the world's total marine biodiversity (Veron et al. 2009).

Perhaps owing to its wide geographic range and habitat connectivity, there is a widespread perception that extinction in the oceans is unlikely (Roberts and Hawkins 1999; Hendriks et al. 2006). Far from this perception, species and marine habitats are increasingly endangered; in fact, in the last decade more than 100 extinctions of marine populations at the local, regional and global scale have been compiled (Dulvy et al. 2003). Although research on biodiversity conservation has increased in recent decades, these efforts are dominated by studies on terrestrial ecosystems, while marine environments remain largely unexplored (Carr et al. 2003; Leslie et al. 2003; Hendriks et al. 2006; Scott et al. 2012). These differences in research turn into differences in conservation measures carried out in each environment. Currently less than 1% of the world's seas are under any form of protected area designation, while protected areas on land cover 11% of the earth's land surface (Naughton-Treves et al. 2005; Toropova et al. 2010). Paradoxically, the fact that oceans are the patrimony of all nations creates a legislation gap and thereby causes a major obstacle to boost the percentage of protected surfaces in the oceans (Norse 2010).

1. 4. COASTAL AND MARINE PROTECTED AREAS

Marine Protected Areas (MPAs) are defined as "areas of the ocean designated to enhance conservation of marine resources" (Lubchenco et al. 2003). MPAs have gained increasing popularity worldwide as tools for biodiversity conservation and ecosystem management. They also help to raise public awareness (Abdulla et al. 2008; Botsford et al. 2009; Guidetti et al. 2013). MPAs have been established with diverse objectives, ranging from conservation of biodiversity, protection of a particular species, groups of species or critical areas (Allison et al. 2003; Micheli et al. 2008), for the prevention of overfishing (Gell and Roberts 2002), and even for the enhancement of local fisheries when MPAs act as suppliers of adult fishes to adjacent non-protected sites (Roberts et al. 2001; Harmelin-Vivien et al. 2008).

Globally, there are over 4000 separate areas designated as MPAs (Wood 2007); however, the actual level of protection, management and regulation within MPAs varies considerably according to its designation (Lubchenco et al. 2003; Al-Abdulrazzak and Trombulak 2012). It is known that implementation of MPAs is only effective if boundaries are drawn to adequately incorporate all important areas used by the protected species, such as foraging and breeding areas (Peckham et al. 2007; Maxwell et al. 2011). However, significant gaps often remain in the design and functioning of MPAs (Pullin et al. 2004; Sale et al. 2005), many of which rarely present an adequate size to preserve a representative sample of regional biodiversity, or to provide adequate protection for species or populations with complex life histories and large area requirements (Gerber et al. 2005).

In developing countries, weak enforcement of regulations relating to protected areas is a common factor. Furthermore, not all protected areas have management plans, and the national authority responsible for their protection is frequently under-funded and under-staffed (Buitrago et al. 2008). For instance, many marine protected areas in the Caribbean suffer from a lack of infrastructure, institutional support, enforceable regulations, and scientific information to guide management policies (Guarderas et al. 2008).

Working in the establishment of MPAs for the protection of marine vertebrate, *a priori* knowledge of spatial habitat-use patterns helps to prioritise area-based protection strategies (Costello et al. 2010; Block et al. 2011). Particularly for marine turtles, we should bear in mind that these species depend as much on marine habitats as on coastal and terrestrial habitats, moving periodically from one to another. Hence, the conservation of these species relies on the adequate protection of these two types of environments. It is obvious that you cannot protect something if you are unaware of its location; this is especially true for the establishment of protected areas for highly migratory species such as marine turtles, whose foraging grounds and suitable breeding habitat are separated by hundreds or thousands of kilometres. In the last decade, our understanding of movements and migrations, the use of preferred areas, and spatio-temporal patterns of habitat use of marine turtles have been widened by the use of satellite telemetry (Godley et al. 2008; Costa et al. 2012). Satellite tracking of marine turtles, as well as in other marine wildlife, has made important contributions to their management and conservation in terms of habitat range and designation of efficient protected areas for them. The effectiveness of extant MPAs in protecting important habitats for marine turtles has been assessed in several recent studies, some of which have underscored the relative effectiveness of these areas since core use areas of many populations largely exceed their boundaries (Maxwell et al. 2011; Nel et al. 2013).

1. 5. MARINE TURTLES

Modern marine turtles belong to an ancient group of reptiles inhabiting the Earth for over 110 million years, since the Cretaceous (Hirayama 1998). From the Cretaceous, four families of marine turtles diverged: Protostegidae; Toxochelyidae; Dermochelyidae; and Cheloniidae, but only the latter two have survived until present (Spotila 2004). Nowadays, marine turtles comprise seven extant species grouped into two families: Dermochelyidae, with the leatherback (*Dermochelys coriacea*) as the single extant species, and Cheloniidae, with six species: hawksbill (*Eretmochelys imbricata)*, Kemp's ridley (*Lepidochelys kempii)*, olive ridley (*Lepidochelys olivacea*), loggerhead (*Caretta caretta*), green (*Chelonia mydas*), and flatback (*Natator depressus*) turtles (Pritchard 1996). With the exception of Kemp's ridley, restricted mainly to the North Atlantic and Gulf of Mexico, and the flatback turtle, endemic to the Australian continental shelf, marine turtles are circumglobally distributed. They inhabit nearly all oceans, occupying unique ecological niches, and exhibiting intra-specific variations in population sizes and trends, as well as reproduction and morphology (Wallace et al. 2011).

1. 5. 1. Life cycle

Marine turtles depend on both marine and terrestrial habitats for their growth and development, from high energy beaches to benthic reefs, and the open waters of the seas. The seven species have similar life cycles (Figure 1.1) with variations in the duration of phases (Miller 1997). Nesting females are philopatric to natal regions with sexually mature animals returning to their natal beaches to breed and nest, and both males and females can be philopatric to breeding areas adjacent to a nesting beach (FitzSimmons et al. 1997; Velez-Zuazo et al. 2008). However, a certain degree of variations in philopatry among populations and species has been described.

In general, female marine turtles typically nest more than once per reproductive season. They do not nest every year and their nesting behaviour is highly stereotypic (Meylan and Meylan 1999). Only females will come ashore to dig a hole into which they deposit between 50-200 soft-shelled eggs, depending on the species (Miller 1997). From six weeks to two months later (depending on the species), hatchlings make their way to the surface of the sand and head to the water. Hatchlings are transported by ocean currents to oceanic habitats, where they live in flotsam, such as Sargassum mats and have an omnivorous diet. Carr (1987) hypothesized that hatchlings spend their first years in oceanic habitats presumably feeding primarily on sea jellies and salps. This period of time is often referred to as the “lost years”; but a recent study based on satellite telemetry methods has shed light on this period (Mansfield et al. 2014). After this oceanic period, they return to coastal waters where they forage and continue to mature. Once adult males and females acquire sufficient resources, they migrate to breeding areas to mate. The time it takes to reach sexual maturity (when they are able to reproduce) varies among species, but ranges between approximately 10-30 years. The distance between feeding and breeding areas can be hundreds, to tens of thousands kilometres, with turtles performing seasonal migrations moving across large expanses of the marine environment.

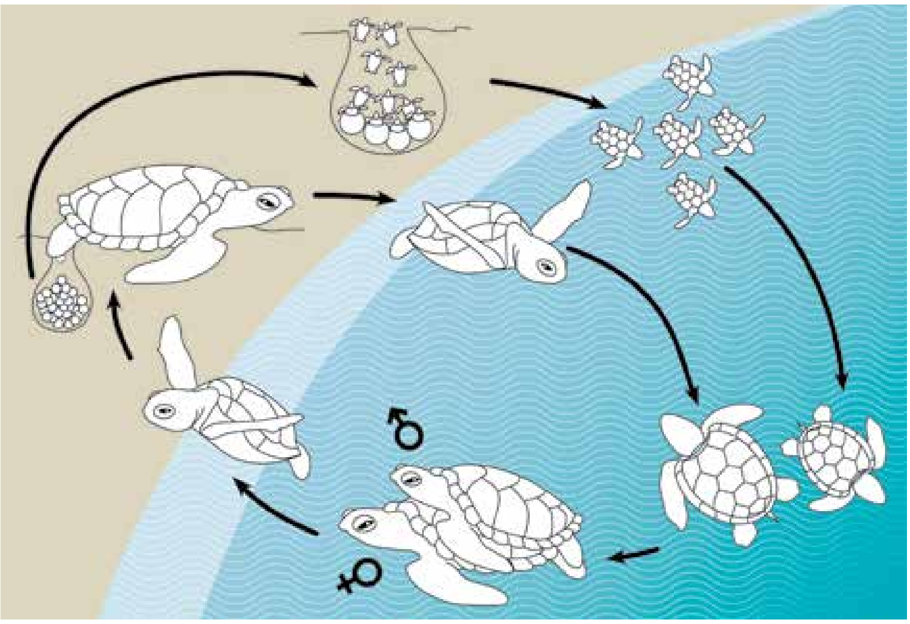

Figure 1.1. Illustration of a generalized life cycle of marine turtles. (Redrawn from Miller 1999, In: The Biology of Sea Turtles, Kennish MJ and Lutz PL eds, CRC Press, Boca Ratón, New York).

1. 5. 2. Roles of marine turtles in ecosystems

Marine turtles are recognised as a "keystone species" because of their ecological impact on their ecosystem structure and function (Bjorndal and Bolten 2003). Marine turtles play fundamental ecological roles in ocean ecosystems by maintaining healthy seagrass beds and coral reefs, providing a key habitat for other marine life and facilitating nutrient cycling from water to land (Bjorndal 2003; Bjorndal and Jackson 2003). Green turtles feed primarily on seagrass (Thayer et al. 1982) and hawksbill turtles have a dietary preference for marine sponges. This can have a positive indirect effect on corals by grazing on coral competitors, and it can affect overall reef benthic biodiversity (León and Bjorndal 2002). The leatherback turtle is a globally significant consumer of jellyfish, playing an important ecological role as a top jellyfish predator (Houghton et al. 2006). Turtles host parasites and pathogens and are substrates for many species of epibionts (Sazima and Grossman 2006; Frick 2013). Moreover, they can improve their nesting beaches by supplying a concentrated source of high-quality nutrients from distant and dispersed foraging grounds (Meylan et al. 1995; Bouchard and Bjorndal 2000).

1. 5. 3. Main threats to marine turtles

Many marine turtle populations have been subject to high levels of harvesting and other indirect threats and all species for which data are available are now of conservation concern (IUCN 2013). Threats vary across regions, but general categories include fisheries bycatch (i.e. incidental capture by marine fisheries operations targeting other species), exploitation of eggs, meat or other turtle products, coastal development, pollution and pathogens, and climate change (Wallace et al. 2011). These threats occur at all stages of their life cycle. Anthropogenic threats such as the slaughter of turtles for their meat and egg take are the main threats at nesting beaches. Other sources of egg mortality are bacterial and fungal infections (Patiño-Martínez et al. 2011), egg predation by ghost crabs, ants and other animal species (Blamires et al. 2003; Caut et al. 2006) or non-natural predators such as feral dogs, pigs, raccoons or mongooses (Ordoñez et al. 2007; Leighton et al. 2008). Eggs and hatchlings can also be threatened by toxic chemicals, such as heavy metals and organochlorine compounds (Roe et al. 2011). Coastal development has affected nesting beaches, with many of these beaches around the world being increasingly developed for human recreation, leading to nesting habitat loss in many places. Once at sea, sharks are the primary natural predator of juvenile and adult marine turtles. Currently, however, bycatch from long-line and trawling activities constitutes the main threat for these species (Wallace et al. 2013 and references therein).

In addition, global climate change could have significant negative effects on the survival of many marine turtle populations. Firstly, the predicted sea level rise could be devastating to many islands, compromising availability of nesting beaches due to extensive coastal flooding, inundation of low-lying coastal areas and heightened coastal erosion (Fish et al. 2008). Secondly, temperature is of profound importance as an environmental factor for marine turtles, since it could alter the intra-annual timing of nesting as well as generate large biases in offspring sex ratio, a critical life history trait (Hawkes et al. 2009; Hulin et al. 2009).

Any threat involving temperature changes is crucial for the survival of marine turtles since, as many reptile species, they exhibit temperature-dependent sex determination (TSD) where primary sex ratio is influenced by the temperature experienced by eggs during incubation (Bull 1980). Phenotypic sex in marine turtles is determined by the temperature prevailing in approximately the middle third of the incubation period (Yntema and Mrosovsky 1982). By definition, the pivotal temperature is the temperature at which both sexes are produced in equal proportions (sex ratio= 1:1). The transitional range of temperatures (TRT) is the range of constant temperatures that yields both sexes in variable proportions (Mrosovsky and Pieau 1991). From this TRT, lower temperatures will produce only males and higher temperatures will produce only females. These parameters are inferred from the artificial incubation of eggs at constant temperatures at the laboratory (Mrosovsky et al. 2009). Where pivotal temperatures are known, incubation temperatures can be used to predict hatchling sex ratios (Mrosovsky et al. 2009). Likewise, differences in nest temperatures manifest themselves as extended or shortened incubation durations (ID), for cooler and warmer conditions respectively, and thus ID can be used to some extent to infer sex hatchling ratios (Mrosovsky et al. 1999). In nesting beaches around the world there is a predominance of beaches that produce female-biased hatchling sex ratios, and some of these biases are greater than 90% female (Wibbels 2003). Climate change can thus lead to a feminisation of some populations, putting them at serious risk (Witt et al. 2010).

1.5.4. Marine turtles conservation and management

In order to protect threatened marine turtle nesting populations, a broad range of conservation programmes have been established around the world in which clutches and hatchlings are protected by moving eggs to enclosed hatcheries or to polystyrene boxes (Pritchard et al. 1993; Brown et al. 2012; Maulany et al. 2012). This practice has the potential risk of altering incubation temperatures and skewing sex ratios, often by cooling the eggs and thus increasing the production of males (Wibbels 2003; Maulany et al. 2012). Likewise, extreme temperature variations can negatively affect the hatching and emergence success of clutches (Özdemir and Türkozan 2006). Since hatcheries may have negative effects on hatching success and natural sex ratio, marine turtle eggs should be incubated in the natural nest at the laying site. It has been proposed that the relocation of eggs to a protected hatchery should be undertaken only as a last resort and only in cases where *in situ* protection is impossible (Mortimer 1999). However, conservation projects in areas hosting highly depleted populations under intense pressure from the illegal poaching of clutches have no other option but to incubate clutches artificially. Lack of funding is also an issue. If these conditions lead to the use of hatcheries as the only conservation strategy, it is essential to assess its effects on hatching success and the resulting sex ratios.

1. 6. MARINE TURTLES IN THE CARIBBEAN REGION

The Wider Caribbean Region consists of twenty-eight sovereign nations and comprises nine ecoregions: Western Caribbean, Southwestern Caribbean, Eastern Caribbean, South Caribbean, Greater Antilles, Bermuda, Bahamian, Southern Gulf of Mexico, and Florida (Spalding et al. 2007). This coastal area is a large marine ecosystem characterised by coral reefs, mangroves, and seagrass beds, but including other environments, such as sandy beaches and rocky shores (Miloslavich et al. 2010) and it is also known for its species diversity and patterns of endemism (Spalding and Kramer 2004). The Caribbean islands encompass a biodiversity hotspot exceptionally important for global biodiversity conservation, due to high levels of species endemism (Roberts et al. 2002). In terms of endemism at the genus level, it ranks third among the world's 34 biodiversity hotspots, with 205 plants and 65 vertebrate genera endemic to the islands (Smith et al. 2004).

There is a complex mix of interacting socio-economic, political, cultural and environmental factors that are driving environmental change and threatening biodiversity in the Caribbean (Anadón-Irizarry et al. 2012). The region has suffered from high levels of habitat loss since the arrival of Europeans in the 1490s. This destruction has reduced the hotspot's original estimated 229,549 km^2 of natural vegetation to just 22,955 km^2 (or just 10%). Nowadays, rising population densities and associated rapid, unchecked coastal development, increasing fishing pressure, agricultural and industrial activities, increased river sediment loading, the introduction of alien species and climate change are identified to be among the major sources of anthropogenic pressure on Caribbean marine life (Burke and Maidens 2004).

Six species of marine turtles occur in the Caribbean: loggerhead turtles, green turtles, hawksbill turtles, Kemp's ridley turtles, olive ridley turtles, and leatherback turtles. Each of these species is classified by the World Conservation Union (IUCN 2013) as either critically endangered (hawksbill and Kemp's ridley), endangered (green and loggerhead turtles) or vulnerable (leatherback and olive ridley). The use of a global extinction risk assessment framework represented by the Red List for the assessment of that species status at the regional scale has been challenged by certain experts (Seminoff and Shanker 2008; Wallace et al. 2011). According to these specialists, this type of assessment will be more useful for conservationists and on-the-ground resource managers. Regardless of the framework used, it's a given that populations of endangered Caribbean marine turtles are far more depleted than previously thought because current conservation assessments do not reflect historical nesting data. In the past, large nesting populations were found on beaches throughout the wider Caribbean (McClenachan et al. 2006) but, nowadays large nesting colonies are rare (Figure 1.2). Nesting grounds receiving more than 1,000 crawls per year range from 0.4% (hawksbill) to 7.0% (Kemp's ridley) of all known species-specific sites. For any species, roughly half of all known nesting sites support fewer than 25 crawls (fewer than 10 reproductively active females) per year

(Dow et al. 2007). Calculations based on historical export data show that modern populations of green and hawksbill turtles are 0.33% and 0.27% of their historical numbers, respectively (McClenachan et al. 2006).

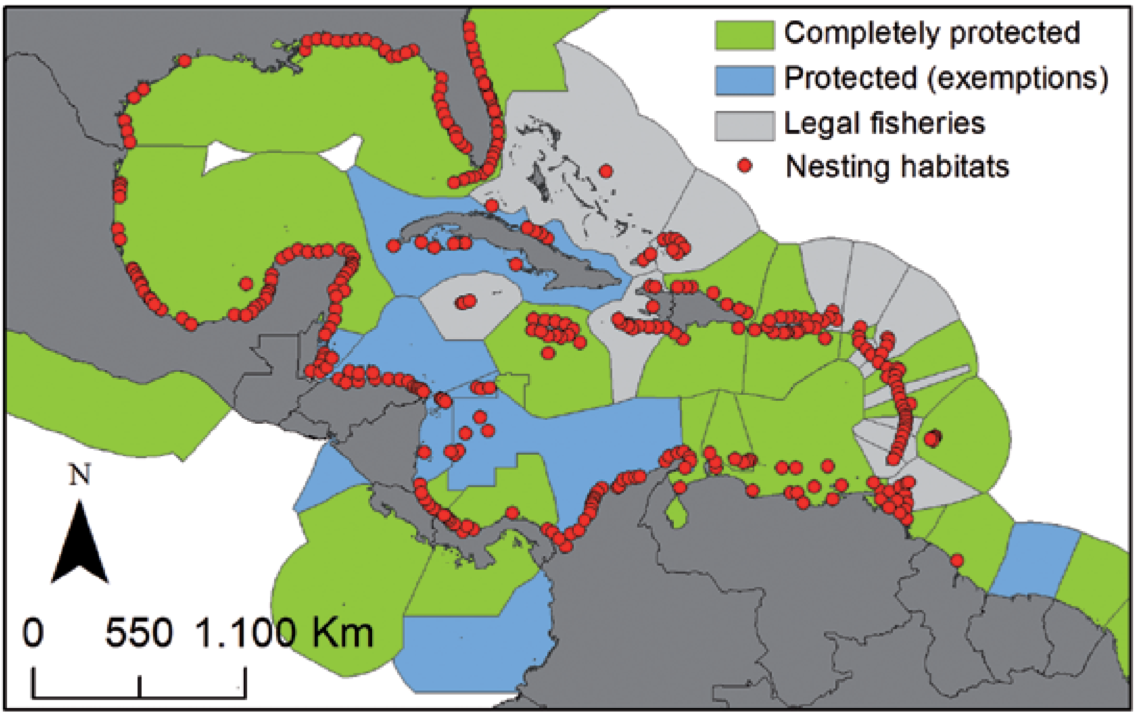

Figure 1.2. Map of nesting habitats (red dots) and legal status of marine turtles within Exclusive Economic Zones (EEZ) for the Caribbean countries (modified from Dow et al. 2007). Blue areas include some protection but allow traditional uses.

Marine turtles in the Caribbean have been subject to exploitation for many centuries, which has resulted in the large-scale reductions in population numbers stated above (Parsons 1962; Eckert 1995; Jackson 1997; Meylan 1999; Bell et al. 2006). Turtle eggs and most turtle body parts (meat, shell and skin) have been valued not only because they provide basic sustenance but also because they have been used to create jewellery for their trade (Fleming 2001; Bräutigam and Eckert 2006). This is particularly dramatic for the hawksbill turtle, sought after for its scutes which have been long valued as raw material for artisans. The intensity of the demand for tortoiseshell products around the world has deeply influenced the survival status of the species (Meylan 1999). In the Caribbean, large-scale commercial harvesting and trade began in the 1950s and 1960s, when international markets for shells from hawksbill and other turtle species expanded (Groombridge and Luxmoore 1989). Despite the restricted international regulations to protect them (all of these species are listed on CITES Appendix I, which prohibits international commercial trade), there is still a large amount of trade in hawksbills products (Bräutigham and Eckert 2006). The Caribbean stands out as a region of exceptionally high levels of tourism development which has led to irreversible environmental degradation (Davenport and Davenport 2006). This development, combined with increasing pollution, sand extraction, sewage and litter have all been detrimental to turtle nesting habitats in the region (Bell et al. 2007).

Based on a recent study carried out to establish conservation priority in different marine turtle Regional Management Units (RMUs) (Wallace et al. 2011), "Low risk-High threats" is currently the most prevalent conservation priority category for marine turtles in the Wider Caribbean region. This category highlights RMUs generally exhibiting large, stable or increasing abundance populations, yet highly under threat. If threats are not abated, these populations could decline in the future, thus warranting intervention before significant population-level impacts can manifest. The leatherback Northwest Atlantic RMU is currently categorised as "Low risk-Low threats" meaning large populations that, in many cases, are well-monitored. This classification supports recent results on marine turtle population status in the region, which quoted that some populations are stable or increasing (Leatherback: Dutton et al. 2005; McGowan et al. 2008; hawksbill: Richardson et al. 2006; Kamel and Delcroix 2009; green: Troëng and Rankin 2005). Although recent conservation efforts have resulted in large population increases at several nesting sites, there are still many gaps on the conservation status of many small, widely dispersed sites lacking intensive population monitoring; many of these unprotected rookeries in this basin are seriously threatened (Lagueux and Campbell 2005; Dow et al. 2007). These populations could play an important role in regional marine turtle conservation and should be considered as potential priorities for management and conservation (Dow and Eckert 2011). In agreement with McClenachan and colleagues (2006), it is better to extend protection across as many beaches as possible so as not to rely on the protection of few exceptional nesting beaches.

1.7. MARINE TURTLE CONSERVATION: THE CASE OF THE DOMINICAN REPUBLIC

The Dominican Republic (DR) with an extension of 48,670 km^2 occupies the eastern portion of the Hispaniola Island, the second largest in size within the Greater Antilles and it is part of one of the most important biodiversity hotspots in Central America. The country supports exceptionally diverse ecosystems, but this goes hand in hand with depletion patterns as observed in wide areas devastated by deforestation and human encroachment (Myers 1988). The country includes a total of 86 protected areas encompassed in six categories with different levels of protection, including one Biosphere Reserve (Jaragua-Bahoruco-Enriquillo) and 19 national parks managed by the National Parks Directorate. Although none of them were created specifically for protection of marine turtles, some provide suitable nesting and foraging habitats.

Four marine turtle species have been reported in the DR's coastal areas: hawksbill turtle, green turtle, loggerhead turtle and leatherback turtle (Ottenwalder 1981).

Hawksbill turtle

The hawksbill turtle is medium size, with a straight carapace length between 60 and 90 cm in the adult stage (Figure 1.3a). The shell is elongated and has overlapping scutes and a serrated edge. This is the most tropical of all marine turtles, with nesting beaches and feeding grounds distributed in tropical and subtropical areas of the Atlantic, Pacific and Indian Oceans between 30° N and 30° S (Baillie and Groombridge 1996). In the western Atlantic Ocean, Gulf of Mexico, and the Caribbean Sea, hawksbills are found from the southern U.S. southward along the Central American coast to Brazil and throughout the Bahamas and the Greater and Lesser Antilles (Meylan and Redlow 2006). Hawksbill turtles forage in a variety of coral and sponge reefs, reef walls, and other hard-bottom habitats throughout the tropics. Sponges are the main component of their diet but they can also include substantial quantities of non-sponge invertebrates that may be present in their feeding habitat (León and Bjordnal 2002). Through their selective foraging behaviour on sponges, hawksbills play a crucial role in the conservation of reef communities favouring reef succession and diversity. Tagging and satellite tracking studies conducted throughout the Caribbean have demonstrated that hawksbills carry out reproductive migrations through the territorial waters of multiple jurisdictions, crossing oceanic areas and moving through other neritic areas to their final foraging grounds (van Dam et al. 2008). Hawksbills nest on both insular and mainland sandy beaches, often in areas with at least some vegetation (Kamel and Mrosovsky 2005). Nesting females typically remain within the vicinity of the nesting beach in the internesting period (Whiting et al. 2006). In foraging grounds juveniles converge from many different nesting aggregations, each consisting of genetically distinct subpopulations (Bass et al. 1996; Velez-Zuazo et al. 2008). Therefore, these are areas of great interest for the conservation of the species.

Leatherback turtle

The leatherback turtle is the largest of all living turtles; adults can reach more than 2 meters in total length and often exceed 500 kg (Figure 1.3b). The lyre-shaped carapace has seven longitudinal ridges, or keels, and black colouration with white spots. The leatherback turtle is an air-breathing diving animal capable of maintaining activity during prolonged dives (with a maximum of 1280 m depth recorded) fuelled by its oxygen stores (López-Mendilaharsu et al. 2009). The leatherback turtle has a worldwide distribution, spreading from tropical to sub-polar oceans and nesting on tropical (rarely subtropical) beaches. Despite its extensive range, distribution is far from uniform and large nesting colonies are rare. In the Western Atlantic, nesting occurs as far north as Assateague Island National Seashore, Maryland, USA (38ºN) (Rabon et al. 2003), and as far south as Torres, Rio Grande do Sul, Brazil (29ºS) (Soto et al. 1997). The largest nesting colonies are located in French Guiana-Suriname (over 40% of the world leatherback population nests, Hilterman and Goverse 2007). Leatherbacks spend the first part of their lives in tropical waters, and once they exceed 100 cm CCL they are considered to be sub-adults and are able to move into cooler waters that have been considered the primary habitat for the species. Leatherback turtles perform long distance migrations (1000s of km) that may span as long as 2-3 years, from nesting beaches to high latitude foraging grounds (Eckert et al. 2006; Hays et al. 2006). In the foraging grounds they feed primarily on soft-bodied animals like jellyfish and pelagic tunicates (Heaslip et al. 2012). In the Atlantic, satellite tracking data has revealed site fidelity for foraging grounds across the North Atlantic (including Canada, the northeastern US and Western Europe), and West Africa (e.g., Ferraroli et al. 2004; Hays et al. 2004; 2006, James et al. 2005; Doyle et al. 2008).

Green turtle

The green turtle (Figure 1.3c) is a circumglobal species occurring throughout tropical and, to a lesser extent, subtropical waters. They are believed to inhabit coastal waters of over 140 countries (Groombridge and Luxmoore 1989). The typical adult has a carapace length reaching 120 cm and can weigh as much as 230 kg in the Atlantic, making this species the largest of the hard-shelled marine turtles (Pritchard and Mortimer 1999). The carapace is broadly oval and the margin is sometimes scalloped but not serrated. Dorsal carapace colour varies among populations, from black to grey-brown spotted patterns. Ventral colour is lighter. The green turtle is the only herbivorous marine turtle foraging primarily on sea grasses in most of its range in large juvenile and adult stages (Hirth 1971), with algae making up the bulk of the diet where seagrasses are lacking (Bjorndal 1985). Green turtles exhibit particularly slow growth rates and the age to maturity for the species appears to be the longest of any marine turtle species (Hirth 1997). This species fills unique ecological roles in seagrass ecosystems by reducing the flux of organic matter and nutrients to sediments. Their removal from this ecosystem would lead to deposition of plant detritus, increase the oxygen demand of sediments, and promote hypoxia (Jackson et al. 2001). The primary green turtle nesting rookeries (i.e. sites with ≥ 500 nesting females per year) are located at Raine Island, Australia (Limpus 2007) and Ascension Island (Mortimer and Carr 1987). In the Caribbean region, the largest remaining green turtle rookeries are located at Tortuguero, Costa Rica and Aves Island, Venezuela (Seminoff 2002).

Loggerhead turtle

The loggerhead turtle is a medium-sized turtle; mature females have a mean straight carapace length of 87 to 105 cm; and a mean weight near 115 kg (Spotila 2004) (Figure 1.3d). They have a characteristic large head with a very strong neck and powerful jaws. The carapace is slightly heart-shaped and reddish-brown in adults and sub-adults, while the plastron (ventral carapace) is generally a pale yellowish colour. Loggerhead turtles are generalists in feeding throughout their lives. In oceanic habitats hatchlings feed on crabs, molluscs, jellyfish and Sargassum (Bjorndal 1997; Bolten 2003). As juveniles, they enter in a benthic feeding stage into estuaries, lagoons and other coastal regions where they consume hard-shelled invertebrates (Dodd 1988; Hopkins-Murphy et al. 2003). However, a dichotomy in feeding strategy has been observed for the species, with juveniles using different habitats for development, thus resulting in adults of the same rookery foraging in coastal waters and the others into oceanic areas (Hatase et al. 2002; Hawkes et al. 2006; Watanabe et al. 2011). Loggerheads are a circumglobal species, occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans, nesting most abundantly in subtropical and temperate areas, and occasionally in the tropics. The world's largest populations of loggerhead turtle nesting colonies are located in Florida (Spotila 2004) and Masirah Island, Oman, in the Indian Ocean (Rees et al. 2010). In the Wider Caribbean, important nesting grounds for this species are located along the southeastern coast of the USA, (mainly in the state of Florida), Brazil, the Yucatan Peninsula in México, Cuba and the Colombian coast (Dow et al. 2007).

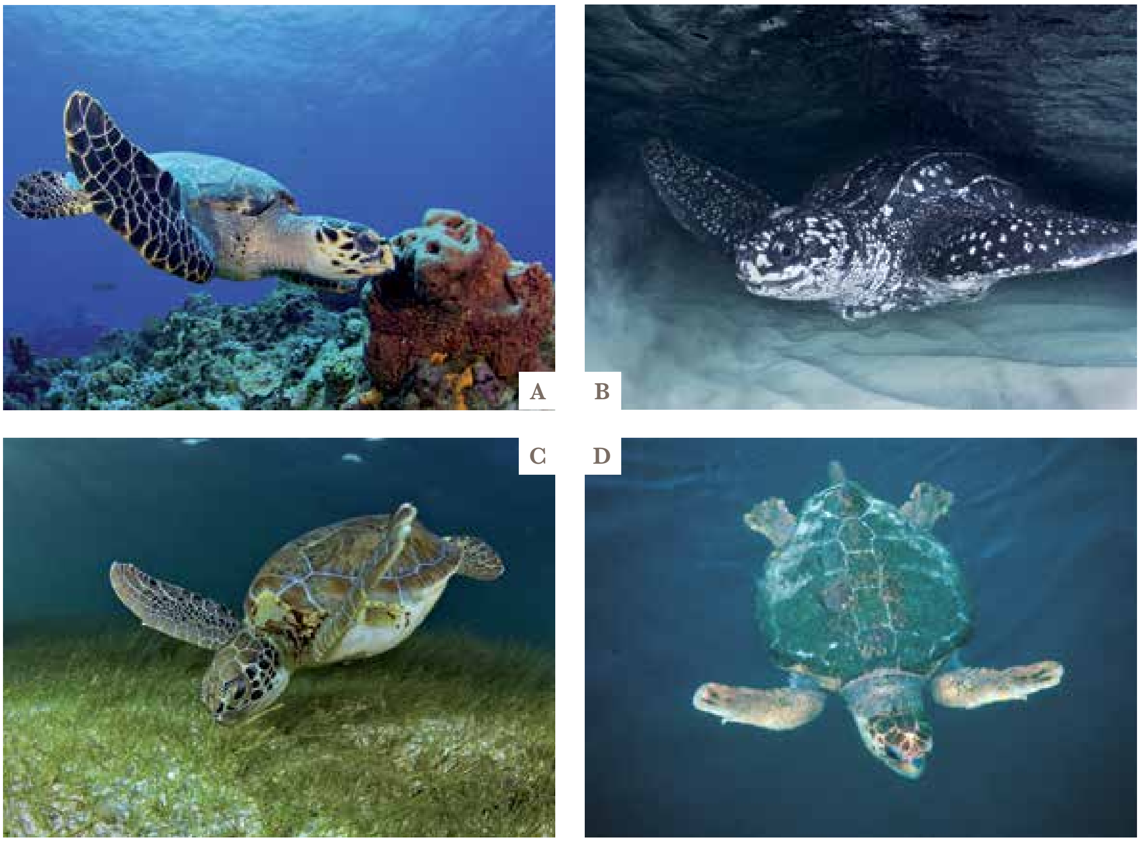

Figure 1.3. Marine turtle species reported in the literature in coastal areas of DR. a) *Eretmochelys imbricata,* b*) Dermochelys coriacea*, c) *Chelonia mydas,* d) *Caretta caretta*. (Photos courtesy of: a, b, c: J. A. Álvarez and d: J. Tomás.

Historically, nesting turtles were abundant in the Dominican Republic (Parsons 1962). Before the arrival of the Europeans to Hispaniola, marine turtles were a prominent element in the existing native cultures (Mota and León 2003). Since the arrival of Columbus, there have been many references about the great presence of marine turtles as well as how appreciated their meat and eggs were (Rodríguez Demorizi 1942; Parsons 1962). The Dominican Republic has a long history of harvesting and commerce in marine turtles, which have constituted an important resource for coastal communities (Ottenwalder 1981; Fleming 2001; Reuter and Allan 2006). The hawksbill's meat, eggs, penis, blood, shell and heart have a known history of use in the Dominican Republic (Figure 1.4a to 1.4e). Green and leatherback's meat, eggs and fat are also consumed. Harvesting practices to supply the Dominican shell industry have represented a serious threat to the species' regional survival outlook. Widespread sales of tortoiseshell items in Santo Domingo and coastal localities have been well documented in shops catering to tourists (Stam and Stam 1992; Dominguez and Villalba 1994; Feliz et al. 2010; Figure 1.4b and 1.4c). Habitat loss due to coastal development constitutes a second important threat to marine turtle nesting habitats (Ottenwalder 1996) (Figure 1.5).

Figure 1.4. Human interaction with marine turtles in Dominican Republic. a) The carapace of a nesting hawksbill captured by humans in Saona Island, b and c) Different types of accessories made from hawksbill shell for sale in gift shops of Santo Domingo (DR), d) Skull and plastron of hawksbill turtle captured by humans in Saona Island, e) Juvenile hawksbill turtle entangled in a fishing net. (Photos courtesy of: a, b and c: Y.M. León, d: J. Tomás and e: S. Aucoin).

Marine turtles are now legally protected and their trade banned in the country by laws dating back to 1966, and recently confirmed through the 2005 Fisheries Law CODOPESCA (Dominican Council of Fisheries and Agriculture). However, the Dominican Republic and other Caribbean islands share a poorly established environmental policy and an institutional weakness (Torres et al. 2000). Therefore, despite the existence of laws and protected areas in the country, "poor supervision" and committed offenses to environmental laws are frequent. Tourism expansion on the coast has led to an appropriation of coastal pieces of land, including territories of coastal and marine protected areas (Heredia 2003). Moreover, tourism constitute a serious threat causing environmental degradation due to solid waste, uncontrolled tourist influx to the beaches or illegal construction on coastal areas (Figures 1.5 and 1.6).

Figure 1.5. a) and b). Debris covering a leatherback nesting beaches at the east of Jaragua National Park (SW DR). (Photos courtesy of: J. Tomás).

Figure 1.6. a) Mass tourism at Bayahibe village (Del Este National Park, SE DR), b) Off-road vehicle at the protected marine turtles nesting beach of Bahía de las Águilas (JNP), and c) Coastal development at Del Este National. (Photos courtesy of: J. Tomás, Y.M. León and O. Revuelta).

Previous knowledge about the conservation status of marine turtle nesting populations in the Dominican Republic relies on the studies carried out by Ottenwalder (1981) and Ross and Ottenwalder (1983) based on surveys along the coast and its small islands. They identified the areas of special interest for nesting leatherback, green and hawksbill turtles. Moreover, they provided the first data on abundance, reproductive biology and conservation status of these species in the DR.

No exhaustive studies on marine turtle nesting population have been carried out during the last three decades; however, there have been numerous reports condemning the alarming level of marine turtle harvesting and commerce in the country (Stam and Stam 1992; Dominguez and Villalba 1994). The lack of updated data on the situation of these species and the threats reported have made it necessary to carry out an exhaustive study in the area not only for the status of Dominican turtles themselves but also for the repercussion that the lack of knowledge could have upon the conservation of marine turtles at the regional level due to the links between populations, the different habitats used by nesting females and their ecological role in Caribbean ecosystems (Bräutigham and Eckert 2006; Dow and Eckert 2011).

1. 8. THE PROJECT

The present study has been carried out within the framework of a five-year project on the study of marine turtle nesting populations in the Dominican Republic. The project was initiated as a pilot study to assess marine turtle populations and the main threats on beaches and coastal waters of the Jaragua National Park. The first studies had the support of a project (PCI-A/2991/05) awarded to the University of Valencia (Spain) by resolution of the Spanish Agency for International Cooperation (AECI) published on 5 January, 2006. During this year, training of local people and public awareness were also included in an attempt to ensure long-term data gathering and conservation legacy. Moreover, possibilities for the development of ecotourism were explored to generate alternative income sources to help prevent direct exploitation of these species. The project was conducted by the Marine Zoology Unit (MZU) of the Cavanilles Institute of Biodiversity and Evolutionary Biology (ICBiBE) with the participation of Dominican institutions, such as the Autonomous University of Santo Domingo (UASD) and the Dominican NGO “Grupo Jaragua”, and with the collaboration of Dominican Ministry of Environment and Protected Areas. This project was renewed in 2007, by granting the same team (PCI-A/5641/06). In the same year, the team of the MZU started the research project CGL2006-02936-BOS (of the Plan Nacional I+D+i, Ministry of Education and Science), that was extended until September 2010. Additional funding has been obtained from the European Union (project IEOST, Marie Curie Intra European Fellowship FP6, and Project RESET-ECO, Marie Curie PEOPLE-2009-RG FP7) and the General Foundation of the University of Valencia. All these projects have been a coordinated joint effort between the University of Valencia, the Dominican NGO Grupo Jaragua, the University of Exeter (United Kingdom), the University of Santo Domingo (through the Research Center for Marine Biology - CIBIMA-and School of Biology), the Technological Institute of Santo Domingo (INTEC), and the Araucaria Project of the Spanish Agency for International Cooperation (AECI), with continuous support from the Dominican Ministry of Environment and Protected Areas. During these five years, all these projects have covered research on reproduction biology and threats (in the present thesis), public awareness and training, but also other studies such as identification of feeding areas of nesting females through satellite telemetry (Hawkes et al. 2012) and populations genetics (Carreras et al. 2013).

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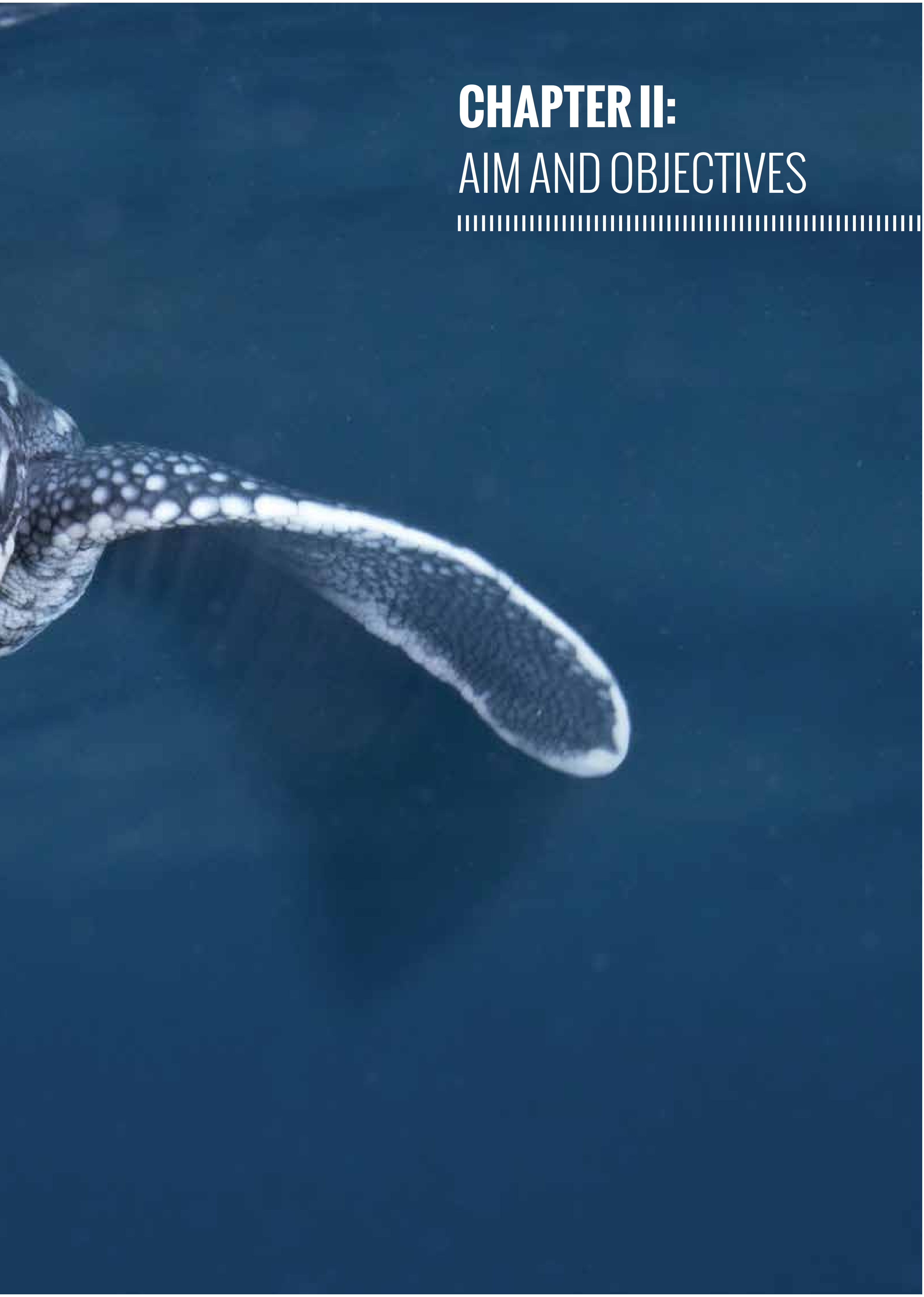
CHAPTER II:

AIM AND OBJECTIVES



AIM

The approach advocated in this thesis is based on the fact that there is a gap of information regarding the conservation status of marine turtle nesting populations in the Dominican Republic. It is recognized that marine turtle conservation requires ample knowledge of different aspects such as reproductive biology or habitat-use patterns. Developing tools and measures for their protection, as well as evaluating the results of such measures are also essential. This thesis is focused on identifying the main marine turtle nesting rookeries in the Dominican Republic, describing the current spatio-temporal patterns of nesting, and assessing the likely impact of the current threats to these nesting stocks, spanning a 5-year study period (2006 - 2010).

OBJECTIVES

The specific objectives of this study are:

1. To identify the main breeding colonies of marine turtles in the Dominican Republic, describing the spatio-temporal patterns of nesting at present and evaluating the impact of current threats to these breeding populations, comparing their status with previous studies in the country.

2. To evaluate the conservation programme based on artificial incubation of eggs, established for the protection of leatherback turtle clutches on the beaches of Jaragua National Park (southwestern area of the Dominican Republic), through the study of the potential effects of this artificial incubation on hatching success and hatchling sex ratio.

3. As in the previous objective, to study the effect of artificial incubation on hatching success and sex ratio to evaluate this measure for the conservation of the hawksbill turtle on Saona Island.

4. To study hatching success and spatial, temporal and reproductive factors affecting it to determine the reproductive value of key beaches of the Jaragua National Park for the leatherback turtle. To highlight the role of the leatherback turtle nesting on threatened beaches of the Jaragua National Park as a vehicle for the conservation of ecosystems and management in this protected area.

5. To investigate the habitat-use patterns of hawksbill turtle nesting in the Dominican Republic during their internesting and foraging periods, identifying core-use areas and comparing them with current marine protected areas in the Caribbean region.

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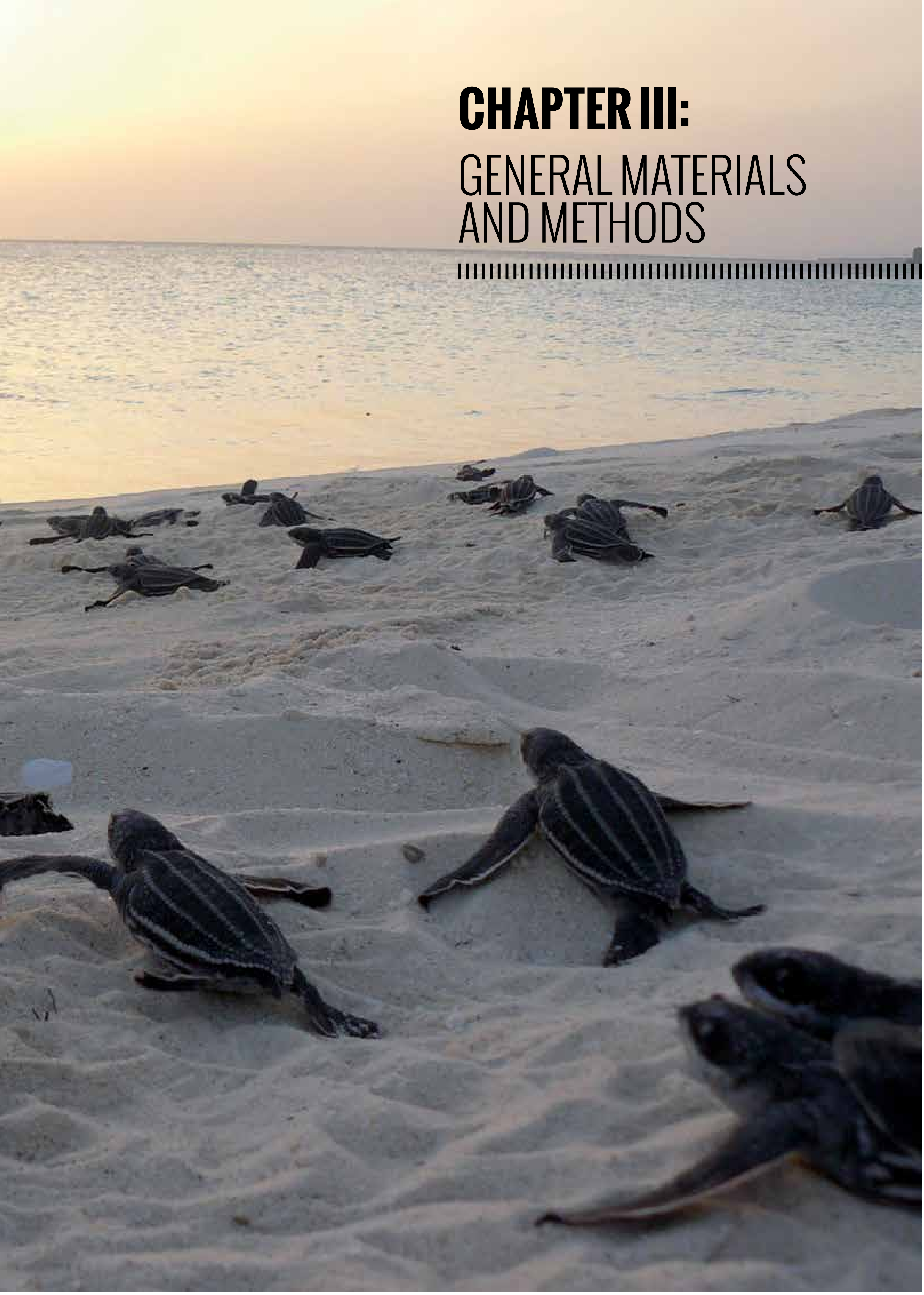

CHAPTER III:

GENERAL MATERIALS AND METHODS

3.1. STUDY AREA

Dominican Republic (DR) shares with Haiti the island of Hispaniola which is part of the Greater Antilles archipelago in the Caribbean region. The country is the second largest Caribbean nation (after Cuba), with 48,445 km^2 and an estimated 10 million people. It has 1,576 km of coastline, including islands, islets and cays; 526 km (33%) on the north coast, 374 km (24%) on the east coast and 675 km (43%) on the south, with a surface of 9,484 km^2 of continental shelf. The country has tropical climate, with a strong maritime influence controlling general weather patterns, apart from the predominant influence of the trade winds. The average temperature is 25 °C (average at coast level: 28 °C) and precipitation ranges from 350 mm to 2.500 mm. Hurricane season in the country runs from June to November. The National System of Protected Areas has 86 units encompassed in six categories: Strictly Protected Areas (n= 8), National Parks (n= 19), Natural Monument (n= 19), Habitat/Species Management Area (n= 13), Strict Nature Reserve (n= 15) and Protected Landscape/ Seascape (n= 12).

Jaragua National Park

Jaragua National Park (JNP) in the southwestern part of the DR covers 1,374 km^2, of which 905 km^2 are marine area (Figure 3.1). The Park includes the islands of Beata and Alto Velo, as well as Los Frailes and Piedra Negra cays. The JNP was established on August 11, 1983 by Presidential Decree and, since 2002, it is one of the core areas of the Jaragua-Bahoruco-Enriquillo Biosphere Reserve. The Park includes numerous ecosystems, ranging from natural forests, beaches, rocky shores, wetlands, seagrasses and coral reefs, with very high levels of flora and fauna endemism. In general, the vegetation is characterized by a great variety of plants adapted to high solar radiation and low precipitation. Its marine ecosystems hosts one the most extensive and best preserved sea grass beds in the southern coast, which supports many threatened species.

The Park comprises two groups of beaches about 50 km apart that provide nesting areas for leatherback (*Dermochelys coriacea*) hawksbill (*Eretmochelys imbricata*) and green (*Chelonia mydas*) marine turtles. At the northeastern side of the Park, on the narrow area of land between Oviedo Lagoon and the Caribbean Sea, are located the beaches of San Luis (11 km length), Mosquea (3.3 km) and Inglesa (1.2 km) (Figure 3.1). These beaches have coarse, dark sand, a steep slope and strong wave action and sea current influence which contributes to the accumulation of large amounts of plastic and other debris on their shores. The westernmost beaches, Bahía de las Águilas (4.4 km) and La Cueva (2.5 km), have fine-grained, coralline, white sand formed by the coral reefs that are nearshore (Figure 3.1). The Park includes other beaches that probably were used for marine turtle nesting in the past, but where currently nesting is scarce.

There are six human settlements with a total population of ca. 5000 located around the boundaries of the Park. Moreover, the Park receives visits of people from the close town of Pedernales and itinerant fishermen either from other parts of DR and from the nearby Haiti. Although these beaches have no infrastructure development, the Park receives 24000 visitors per year, most of them visiting the western beaches. In recent years, the tourism development of Bahía de las Águilas has been a source of debate on whether mass tourism and mining should be established to enhance the economy of the region (Wielgus et al. 2010).

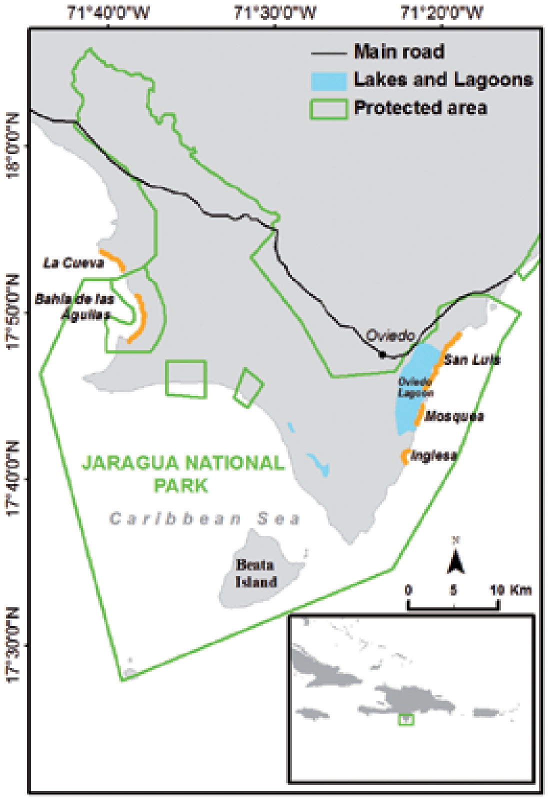

Figure 3.1. Jaragua National Park limits (green line), indicating the beaches of La Cueva and Bahía de las Águilas in the west and the beaches of San Luis, Mosquea and Inglesa in the east (orange lines). The insets indicate the location of the main maps in the Caribbean. Green areas inside the Park are designated as national recreation areas.

Saona Island

Saona Island is included in the Del Este National Park (DENP, south-east DR), which was founded on September 16, 1975. With an area of 110 km^2, Saona is the largest island adjacent to the DR (Figure 3.2). For the most part, beach vegetation is dominated by coconut (*Cocos nucifera*) plantations, sea purslane (*Sesuvium portulacastrum*), sea rosemary (*Suriana maritima*), sea grape (*Coccoloba uvifera*), goat's foot creeper (*Ipomoea pes-caprae*) and native grasses. The neritic sea adjacent to the nesting beach is composed of coral reefs and seagrass beds which provide prime habitat for hundreds of species of plants, birds, fish and other marine animals. Saona Island is also home for several endemic, threatened, or endangered plant and animal species. There is an uncontrolled access to the park by fishermen from other parts of the country and expanding tourism industry that increases pressure on coral reefs and other natural resources.

There is one permanent human settlement in Saona, Mano Juan village, with a population of c. 300 inhabitants.

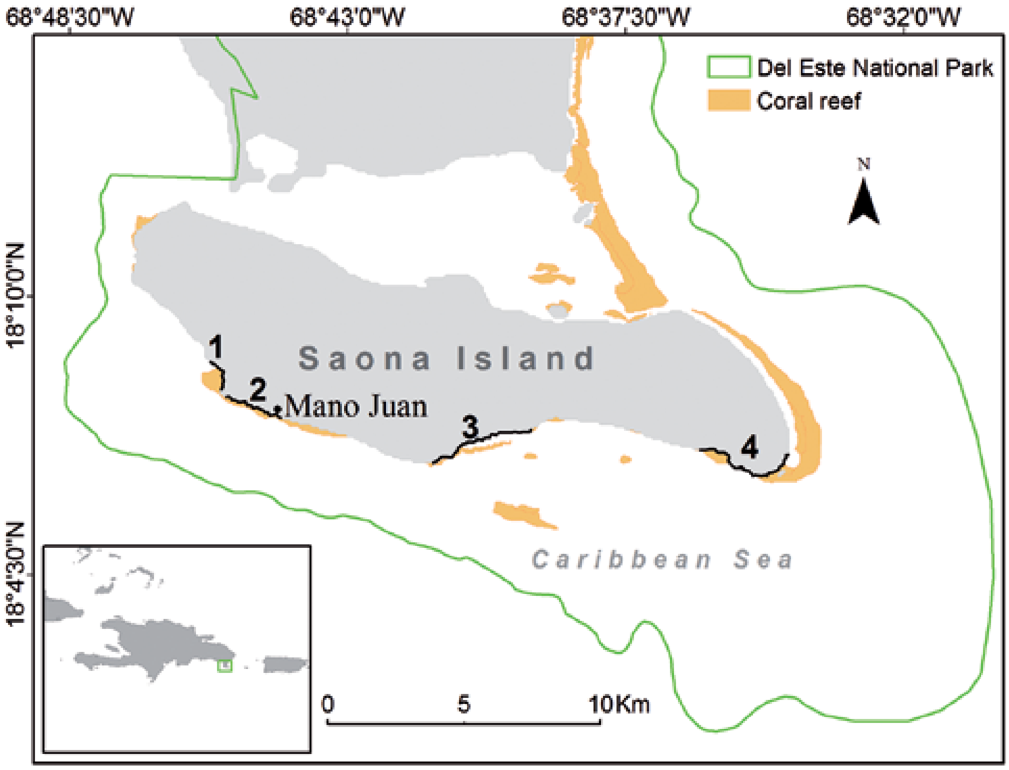

Figure 3.2. Saona Island, indicating Mano Juan village and the four surveyed areas (black lines) of (1) El Toro, (2) Mano Juan, (3) Canto la playa and (4) Faro Punta Cana. The insets indicate the location of the main maps in the Caribbean.

3. 2. BEACH SURVEYS

In order to appropriately design nesting beach surveys, previous knowledge on nesting activity and nesting areas (Ottenwalder 1981) was revised and, interviews to rangers and local fishermen were conducted to determine the most important beaches to survey and the period of the nesting season.

Terrestrial surveys allow counting turtle's tracks and species identification. Daytime surveys were undertaken on foot by researchers and trained local people on the beaches of the JNP and Saona Island during five entire consecutive nesting seasons, 2006-2010. From 2006 to 2008, eastern beaches of the JNP were monitored once a week, except for Inglesa beach where surveys were carried out every two weeks due to the distance and difficult access to that beach. From 2009 onwards, these beaches were daily surveyed by JNP's rangers trained by the research team. On the western beaches of the Park 3-4 daytime surveys were carried out per week, but survey effort ensured the record of all nesting events every year. Other remote beaches at the JNP with low levels of nesting previously reported were also occasionally visited to confirm reports of nests from reliable informants. In Saona Island, beaches were patrolled at least once per week throughout the year, but in 2008 survey effort was increased to 3-4 surveys per week during the period of peak nesting activity (June to November, see chapter 4).

Surveys in other areas of the Dominican Republic were undertaken during 2006-2010. Visits, sporadic patrols and interviews were made in 11 areas of the northern coast in 2006, 2007 and 2008, and on six beaches of the eastern and southern coasts in 2008, 2009 and 2010 (Figure 3.3). Reports about other nesting events, particularly from resorts or beach front developments, were also compiled.

During daytime surveys on the JNP and Saona Island, we patrolled the beaches by foot to detect all recent tracks of nesting females (Figure 3.4a and 3.4b). We recorded total number of emergences of nesting females, identified the species based upon track characteristics (Schroeder and Murphy 1999; Figure 3.4c and 3.4d) and confirmed whether each activity had successfully resulted in clutch deposition, by

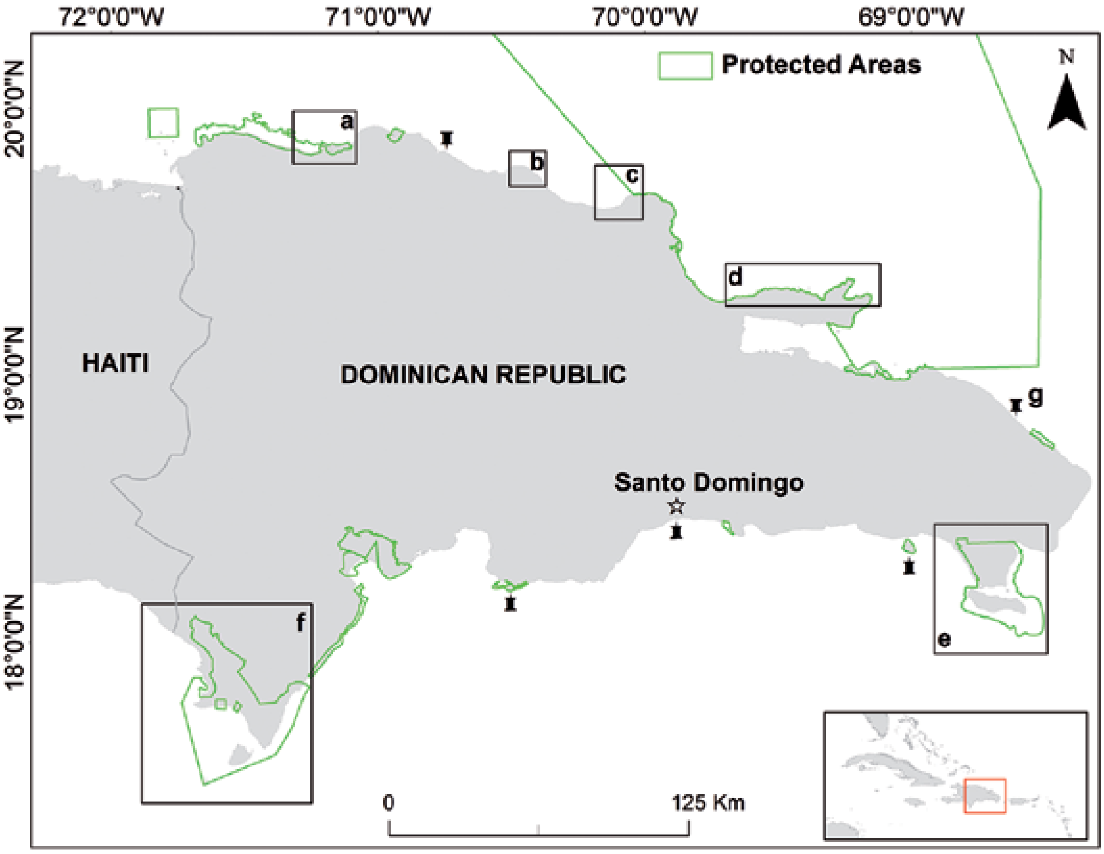

Figure 3.3. Map of the Dominican Republic indicating places where surveys and interviews have been undertaken. a-d: beaches visited with interviews performed and sporadic nesting recorded, e: Del Este National Park with Saona Island, f: Jaragua National Park, g: Del Muerto beach, also surveyed sporadically during the study period (see text for details). Black pushpins indicate other records of low level of sea turtle nesting activity communicated by others and/or occasionally visited by members of the project team and collaborators.

digging to find the clutch. Nest location was recorded using a global positioning system (GPS) and at-site features, and turtle tracks were marked to avoid duplicate counts (Figure 3.5a). Signs of eggs taken by people were recorded (presence of probing sticks, evidence of digging and broken eggshells, and/or human footprints around the nesting site; Troëng et al. 2004) (Figure 3.5b and 3.5c). Remaining clutches were either camouflaged and incubated under natural conditions in the beach or, when clutches were in high risk of subsequent take (because of proximity to a human settlement or presence of people in the area), they were transferred to protected places for egg incubation in boxes.

Because of funding limitations, night surveys were only carried out on Saona Island and the western beaches of the JNP. During night surveys, when finding a female turtle nesting the protocol was to leave someone from the survey team with the animal until she finished the eggs laying, while the rest of the team continued with the survey. After oviposition, turtles were measured, tagged and examined for possible injuries, epibionts or other abnormalities (see next section).

Figure 3.4. Different images of the surveys carried out during the study period (2006-2010). a) nesting beaches at Saona Island, b) daytime surveys on Bahía de las Águilas (western JNP), c) data collection of a leatherback turtle track at Bahía de las Águilas beach, d) hawksbill turtle track on Canto de la playa beach (Saona Island). (Photos courtesy of: Y.M. León, O. Revuelta, J. Tomás and A. Mason).

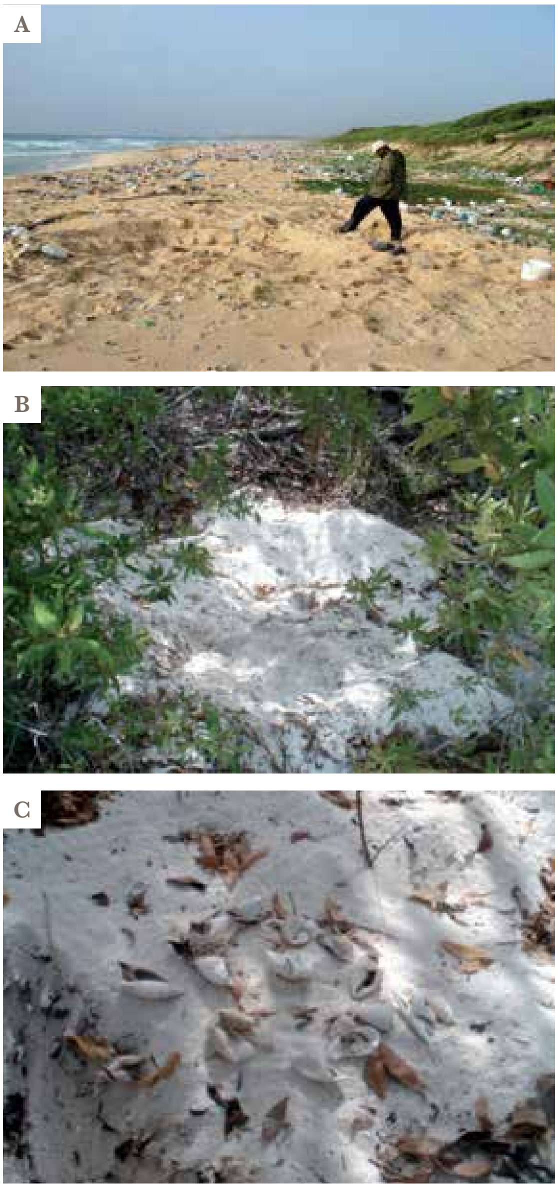

Figure 3.5. a) Leatherback nest surrounded by debris on Mosquea beach (eastern JNP), b) predated hawksbill clutch in Saona Island, c) rests of eggshells from a predated clutch. (Photos courtesy of: P. Feliz and Y.M. León).

3.3. DATA COLLECTION AND CLUTCH RELOCATION

For every nest, we collected data on date and time of laying, GPS coordinates, nest location across the beach (distance to high tide line), nest location along the beach (sector) and beach zone (open sand, vegetation border, and within vegetation). A flexible tape was used to obtain curved measures of the turtles; curved carapace length (CCL) and maximum curved carapace width (CCWmax). Hawksbill and green turtle carapaces were measured from the anterior point at midline (nuchal scute) to the posterior notch at midline between the supracaudals (Figure 3.6a); leatherbacks were measured from the anterior edge of the carapace at the midline to the posterior tip of the caudal peduncle. Carapace width was measured at the widest point for the three species (Figure 3.6b). Hawksbill and green turtle females were tagged with small metal Inconel tags on the trailing edge of both fore-flippers (Figure 3.6c). Leatherback females were tagged with large metal Inconel tags (National Brand and Tag Co., Newport, USA) fitted between the tail and rear flippers (Figure 3.6d).

Figure 3.6. Measuring and tagging marine turtles. a) measuring CCL of a nesting hawksbill turtle in Bahía de las Águilas, b) measuring CCW of a nesting leatherback turtle at La Cueva beach, c) detail of tagging a nesting hawksbill turtle, and d) tagging a nesting leatherback turtle between the tail and right rear flipper at western JNP. (Photos courtesy of: Y.M. León, O. Revuelta, J. Tomás and Grupo Jaragua).

Once the turtle left the beach we proceeded to the management of the clutch. The fate of the clutches was decided according to the risk of flooding, presence of natural/feral predators or passers-by, and its proximity to fishing camps and villages. If no threats were found, clutches were left for incubation in the beach under natural conditions (*in situ*). In those cases with clear risk of clutch predation, the nest was excavated and eggs transferred to polyethylene exterior boxes with polyurethane foam filling (dimensions 30 cm wide x 50 cm long x 32 cm deep) for their incubation at the park rangers' huts (Figure 3.7a and 3.7b). Two to three centimetres of nest sand was put at the bottom and sides of the box to prevent contact of eggs with walls. Clutches were carefully excavated by hand and eggs extracted avoiding their rotation, as has been recommended to improve the success of delayed relocations (Abella et al. 2007; Figure 3.7b). Once in the box, the top eggs were also covered with two to three centimetres of sand proceeding from the original nest chamber.

For each clutch the following data were collected (Figure 3.7c to 3.7e):

- **Clutch size (number of yolked eggs):** the number of eggs laid into the nest, excluding yolkless eggs.
- **Number of yolkless eggs:** small eggs (around half the diameter of yolked eggs) containing mostly albumen encapsulated by a shell.
- **Egg size:** Ten eggs were chosen at random from each clutch. Each egg was cleaned of adhering sand and the greatest diameter was measured to the nearest 0.1 cm with a caliper.

- **Egg weight:** Eggs were weighed using an electronic balance read to a minimum accuracy of 0.1 g.
- **Nest bottom depth:** Depth from beach surface to bottom of egg chamber (cm).

Each box was labelled with a code indicating beach name, laying date and number of eggs incubated. The boxes were stored in Park rangers' huts (Figure 3.8a and 3.8b) for their incubation and were checked daily throughout the incubation duration and lids opened for two to three hours a day to allow air circulation.

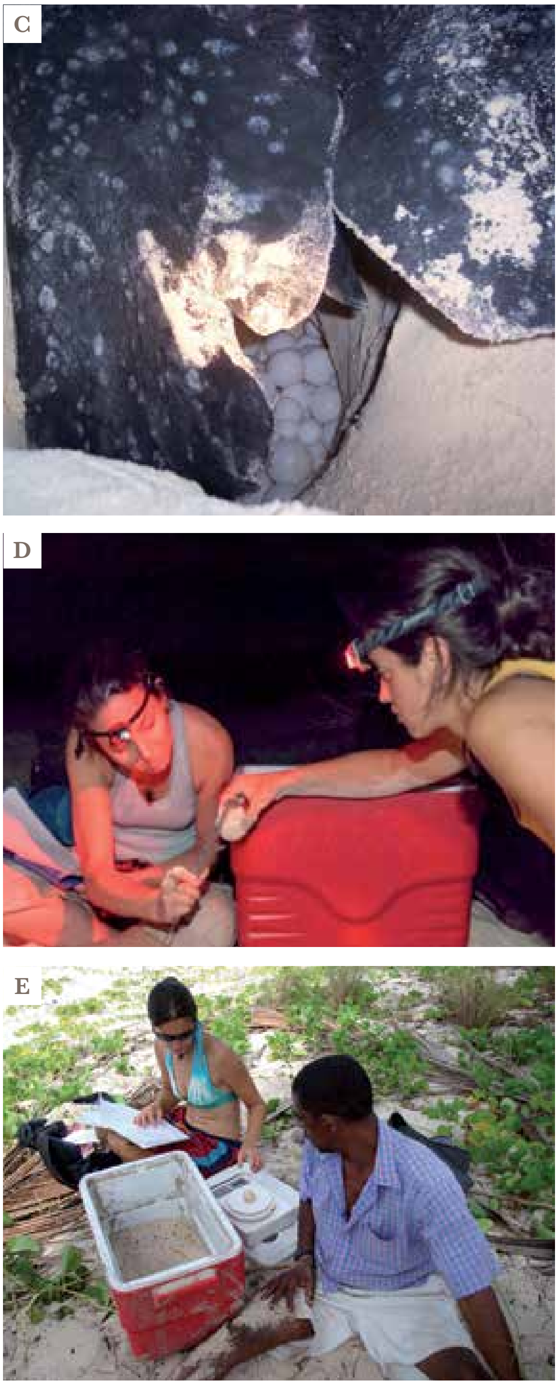

Figure 3.7. Images of egg collection. a) and b) Counting and transferring hawksbill eggs to a polyethylene box in Saona Island, c) Yolked and yolkless eggs in a nest chamber of a nesting leatherback turtle, d) measuring leatherback turtle eggs, e) weighing hawksbill turtle eggs. (Photos courtesy of: Y.M. León, O. Revuelta and J. Tomás).

Figure 3.8. Rangers huts where artificial incubation was carried out. a) Saona Island and b) Bahía de las Águilas. (Photos courtesy of: Y.M. León and J. Tomás).

3. 4. HATCHING SUCCESS AND EMERGENCE SUCCESS

Once artificially incubated eggs hatched, clutches were carefully excavated no earlier than 48 hours after the last sign of hatchling emergency. Hatchlings were released to the sea on their origin beaches (when possible) when they entered into frenzy. Before they were released, random samples of 20 hatchlings from each clutch were measured (straight carapace length, SCL) to the nearest 0.1 cm with a calliper, and weighed to the nearest 0.1 g with an electronic scale. Clutches left incubating on the beach (*in situ*) were monitored during the incubation duration and, when hatchling tracks were observed out of the nest rangers marked them. Clutches were excavated for hatching success study no earlier than 48 hours after track detection.

During the study of both *in situ* and artificially incubated clutches the next categories were considered and quantified as follows (Figure 3.9):

- **Live hatchlings:** hatchlings found alive among shells.
- **Dead hatchlings:** hatchlings found dead out of the shell.
- **Yolkless eggs**: number of eggs without yolk.
- **Egg shells:** empty open shells remaining after hatchlings emerge from the nest. We counted as one shell those that make up more than 50% of the egg size.
- **Unhatched eggs:** Eggs with sign of embryo development. These eggs were opened and classified as:

Early stage embryonic death: eggs that had partial calcification of the shell or evidence of an early stage dead embryo (Bell et al. 2003).

Late stage embryonic death: eggs that contained dead embryos at late stage of development.

Figure 3.9. Clutch study. a) An exposed leatherback hatched clutch contents: 1. Yolkless eggs, 2. Yolked eggs, 3. Hatched eggshells. From b to e) Excavating and collecting data of leatherback and hawksbill clutches at study sites. (Photos courtesy of: a, b: J. Tomás, c: Héctor González, d and e: O. Revuelta).

For the study, we defined hatching success and emergence success as follows, according to the literature (Miller 1999):

Hatching success (HS) is the number of hatchlings that hatch out of their egg shell (number of empty egg shells in the nest). HS was calculated as:

$$\text{HS} = \left(\frac{\textit{Number of egg shells}}{\text{egg shells + unhatched eggs}} \right) \text{x } 100$$

Emergence success (ES) is the number of hatchlings that reach the beach surface. ES was calculated as:

$$\text{ES} = \left(\frac{\textit{Number of egg shells - (live hatchlings + dead hatchlings)}}{\text{egg shells + unhatched eggs}} \right) \text{x } 100$$

Hatchlings were released to the sea on their origin beaches (Figure 3.10).

Figure 3.10. Releasing leatherback and hawksbill turtle hatchlings in Bahía de las Águilas and Saona Island. (Photos courtesy of: R. Briones, Y.M. León, P. Feliz, J. Tomás and O. Revuelta).

3. 5. ESTIMATING HATCHLING SEX RATIO

Since the sex of marine turtle hatchlings can not be assessed from external morphology and scarifying hatchlings was not an option from an ethical and conservation perspective, incubation duration of clutches was used as an indirect method to estimate hatchling sex ratio (Mrosovsky et al. 1999). For each clutch, we calculated the incubation duration (defined as the number of days between egg laying and the first hatchling emergence). This method has been devised for situations in which only pivotal incubation duration (that duration giving 50% of each sex) is available from laboratory studies, and no field samples have been collected and sexed. The laboratory studies start with the incubation of eggs at various constant temperatures, followed by calculation of pivotal incubation duration and a curve relating duration to sex ratio (Godley et al. 2001). Then this laboratory curve has to be adjusted to a curve appropriate to the field data by adding the hatch-emergence interval to the incubation durations. Finally, the adjusted curve is used for converting values obtained in the field on incubation durations (duration from laying to hatchling first emergence) to the corresponding values for clutch sex ratio estimations. Since no suitable data from laboratory are available for marine turtles from DR nesting beaches, we estimated hatchlings sex ratios using the conversion curve relating incubation duration to hatchling sex ratio derived from artificially incubated eggs originating from Suriname (Godfrey 1997) and from Mona Island, Puerto Rico (Mrosovsky et al. 2009) to estimate leatherback hatchling sex ratio in JNP and hawksbill hatchling sex ratio in Saona respectively. To apply these curves to artificially incubated clutches, we added 4.1 days to our incubation duration data, which is the only published estimation of the time gap between hatching and emergence of hatchlings at the sand surface (Godfrey and Mrosovsky 1997).

3. 6. TRACKING HAWKSBILL TURTLES

Platform Terminal Transmitters (PTT) were attached on nesting hawksbill in Saona Island (n = 8) and JNP (n = 1). To prepare turtles for satellite tag attachment, a portable wooden corral was erected around each turtle following nesting. The carapace of each turtle was prepared by scrubbing to remove epibionts, sanding lightly, and cleaning with acetone, and a PTT was attached with 2-part epoxy (based on the methods of Blumenthal et al. 2006; Figure 3.11).

Turtle positions were determined with the ARGOS system which assigns location accuracy estimates (location class [LC]) to each reported location that are classified as 3, 2, 1, 0, A, B, or Z. Argos assigns accuracy estimates of <250 m for LC 3, 250 to <500 m for LC2, 500 to <1500 m for LC1, and >1500 m for LC0 (CLS, 2011). The estimated accuracy is unknown for LCs A and B, and locations failing the Argos plausibility tests are tagged as class LCZ. Tracking and remote sensing data were downloaded and filtered using the Satellite Tracking and Analysis Tool (STAT) (Coyne and Godley 2005) program available from http://seaturtle.org. Locations that were used for assessing movements and delineating core-use areas in this study were taken from Argos Location Classes 3, 2, 1, A and B (shown to be the best locations to describe hawksbill movements; Gaos et al. 2012). Biologically unreasonable results of location points were also filtered, such as unlikely swimming speeds (>5 km h^{-1}; see Luschi et al. 1998), turning behaviour (<25° turn angle) or erroneous locations on land.

Home range was estimated using the minimum convex polygons (MCP), a non-statistical measure which encapsulates the area used by an individual within a polygon formed by joining the outer-most sighting positions (Burt 1943). MCP is a simple calculation that allows for comparisons between studies (Hooge et al. 1999), however is unable to define fine-scale movements within the polygon. Hence, core-use areas of activity were identified using fixed kernel density estimation (KDE) with individual kernel contours delineated using a smoothing factor (h) with the spatial analyst extension of ArcGIS (ESRI). We used a 90% KDE to represent the overall home range of a turtle and a 50% KDE to represent the core area of activity (Powell 2000). To analyze the location of turtles with respect to marine protected areas (MPA), MPAs boundaries were overlaid on all resulting maps and summed location data with respect to the boundaries. Site fidelity for the females was quantified using a residency index (Mason and Lowe 2010) calculated by dividing the number of days a female was detected within the DENP's boundary by the number of days the female was monitored in the area (i.e. internesting period).

Bathymetric data were sampled from the General Bathymetric Chart of the Oceans GEBCO 1-Minute Global Bathymetry Grid (www.bodc.ac.uk/projects/international/gebco/gebco_digital_atlas). MPAs boundaries were downloaded from the World Database on Protected Areas (www.wdpa.org).

Figure 3.11. Attaching satellite transmitters on nesting hawksbill turtles at Saona Island. (Photos courtesy of: O. Revuelta, J. Blumenthal and H. González).

3. 7. STATISTICAL ANALYSES

In chapters 5 we analyzed data using Generalized Linear Mixed Models (GLMMs) that were performed with the lme4 package (Bates et al. 2008) in R (version 2.14.0) and the graphical output was produced with the sciplot package (Morales 2012). In chapter 6, statistical analyses were carried out using the statistical package SPSS v17 (IBM). In chapter 7, analyses were performed using the lnme package (Pinheiro et al. 2011) in R (version 2.14.0) for the linear mixed models. In both chapters, model selection was based on ΔAIC values lower than 2, calculated as the difference between the AIC values for each model and the model with lowest AIC, and model weights (wi) (Burnham and Anderson 2002). Data of chapter 8 were mapped in ArcGIS 10.0 (ESRI, 2010). Specific analyses developed for each study, or other statistical programs used, will be indicated in each chapter. Statistical significance was set at $P < 0.05$, unless otherwise stated.

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CHAPTER IV:

PROTECTED AREAS HOST IMPORTANT REMNANTS OF MARINE TURTLE NESTING STOCKS IN THE DOMINICAN REPUBLIC

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ABSTRACT

Nesting by marine turtles in the Caribbean has declined considerably, mainly because of human exploitation, but there has previously been no monitoring in the Dominican Republic. We present the first detailed assessment of the status of marine turtle nesting in the country, based on surveys during 2006-2010. Nesting populations of hawksbill *Eretmochelys imbricata* and leatherback turtle *Dermochelys coriacea* are of regional importance and the green turtle *Chelonia mydas* is still present, although nesting in low numbers. The two main nesting sites are within protected areas: the Jaragua National Park in the south-west, important for leatherback turtles (mean of 126 nests per season), and Del Este National Park on Saona Island in the south-east, principally for hawksbill turtles (mean of 100 nests per season). Comparison with historical data suggests all rookeries are profoundly reduced in size. Although the main nesting beaches are within protected areas, illegal egg-take and meat consumption continues there and also elsewhere in the country.

INTRODUCTION

Many marine turtle nesting rookeries in the Caribbean have been reduced or extirpated by human exploitation (Parsons 1962; Bjorndal and Jackson 2003; Bell et al. 2006; Bräutigam and Eckert 2006). Although a number of rookeries have been studied for several decades and their status is well documented (Dutton et al. 2005; Troëng and Rankin 2005; Richardson et al. 2006; Diez and van Dam 2007; Beggs et al. 2007), many other Caribbean rookeries remain poorly described (McClenachan et al. 2006; Dow et al. 2007). The Dominican Republic is an area where information on marine turtle nesting activity is scarce and outdated. Unlike foraging habitats, which have been studied since 1996 (León and Diez 1999; León and Bjorndal 2002), the conservation status of nesting populations in the Dominican Republic has never been comprehensively assessed, although there have been suggestions that they are seriously threatened (Ottenwalder 1981, 1987; Stam and Stam 1992; Dow et al. 2007).

The Dominican Republic has a long history of harvest of and commerce in marine turtles, which have constituted an important resource for coastal communities (Ottenwalder 1981; Fleming 2001; Mota and León 2003; Reuter and Allan 2006). Marine turtles are now legally protected and their trade banned in the country by laws dating from 1966 and recently confirmed through the 2005 Fisheries Law CODOPESCA (Dominican Council of Fisheries and Agriculture). Slaughter of leatherback turtles *Dermochelys coriacea* for their meat and eggs was widely recorded in the 1980s (Ottenwalder 1981; Ross and Ottenwalder 1983) and juvenile and adult green *Chelonia mydas* and hawksbill turtles *Eretmochelys imbricata* were exploited for their meat, eggs, shell and other products (Ottenwalder 1981; Dominguez and Villalba 1994; Mota and León 2003). By the 1980s it was estimated that 1,000-2,000 turtles were taken annually, of which 70% were green and hawksbill turtles and 30% loggerhead *Caretta caretta* and leatherback turtles (Ottenwalder 1996). Most of these turtles were caught by divers or taken incidentally in seine nets but many were captured from beaches whilst nesting. Despite legislation banning trade in tortoiseshell, an estimated 600 kg of hawksbill shell were used annually in the Dominican Republic in the 1980s (Ottenwalder 1987). Between 1970 and 1986, Japanese customs data indicate that a total of 4,366 kg of hawksbill shell were exported to Japan (Fleming 2001). Domestically, sale of items made from tortoiseshell has been reported more recently (Mota and León 2003; Reuter and Allan 2006; Feliz et al. 2010).

The Dominican Republic receives four million visitors annually, and this mass tourism has resulted in degradation of coastal habitats (León 2004; Wielgus et al. 2010). Between 1980 and 1996 seven major coastal areas, including a large proportion of the most important turtle nesting habitat, were developed (Ottenwalder 1996). By the 1980s six marine protected areas had been established, four of which (Montecristi, Del Este, La Caleta and Jaragua National Parks) cover c. 22% of the coastline. There are nine coastal parks managed by the National Parks Directorate and, although none of them were created specifically for protection of marine turtles, some provide nesting and foraging habitat for them. However, there is little information available on marine turtles nesting in these areas.

Surveys in the 1970s and 1980s previously constituted the main reference on the status of marine turtles in the Dominican Republic (Ottenwalder 1981; Ross and Ottenwalder 1983). Based on a combination of five countrywide aerial surveys conducted in April-July 1980, interviews with fisherman and local people, and nonintensive beach surveys, Ottenwalder (1981) estimated 420 (range 240-600) hawksbill, 380 (range 253-507) leatherback, 260 (range 160-360) green and 60 (range 30-90) loggerhead turtles nesting annually. Ross and Ottenwalder (1983) identified four areas of special interest for nesting leatherback turtles: the beaches of San Luis and Bahía de las Águilas in Jaragua National Park in the south-west and Del Muerto and Macao beaches on the east coast. More recently leatherback turtle nesting has been recorded on the eastern beaches of Jaragua National Park during irregular beach walks during April-June (Dominici 1996).

The lack of comprehensive studies and recent information and the threatened status of marine turtles in the Dominican Republic necessitated an updated assessment to help target effective conservation action. Our study had three main objectives: (1) to present the first systematic assessment in 30 years of the status of marine turtles nesting in the entire Dominican Republic based on 5 years (2006-2010) of systematic surveys, (2) to identify the main nesting rookeries and describe the current spatio-temporal patterns of nesting, and (3) to assess the likely impact of the current threats to these nesting stocks.

MATERIALS AND METHODS

Study area

The Dominican Republic is in the eastern part of the Caribbean island of Hispaniola, which it shares with Haiti. It has 1,389 km of shoreline of which c. 800 km are sandy beaches. In February 2006 beach surveys and interviews with local people were carried out on 31 beaches previously described as important nesting sites (Ottenwalder 1981; Ross and Ottenwalder 1983). Following these surveys intensive sampling concentrated in the two areas where nesting still seemed to be significant: Jaragua National Park and Saona Island (Figure 4.1). Opportunistic surveys were conducted in other areas during 2006-2010 to detect any other potential nesting sites and to verify reports received during the study (Figure 4.2).

Jaragua National Park in the south-west covers 1,374 km^2, of which 905 km^2 comprises beaches and dunes. The westernmost beaches, Bahía de las Águilas (4.4 km long) and La Cueva (2.5 km), have fine-grained, coralline, white sand. Although these beaches have no infrastructure development, they receive >24,000 visitors per year (Wielgus et al. 2010). The easternmost beaches of San Luis (11 km), Mosquea (3.3 km) and Inglesa (1.2 km) are on the narrow area of land between Oviedo Lagoon and the Caribbean Sea (Figure 4.1). Eastern beaches are wider than those in the west and have coarser, darker sand, a steeper slope and strong wave action, which contributes to the accumulation of large amounts of plastic and other debris on the beaches of San Luis and Mosquea (OR, YML, PF and JT, pers. obs.). Six human settlements with a total population of c. 15,000 are located around the boundary of the Park.

Saona Island (a part of Del Este National Park) in the south-east has an area of 110 km^2. The main nesting zones for marine turtles include 12 narrow, white sand beaches (15 km long in total) interspersed with rocky areas on the south and west of the island (Figure 4.1). Access to some of these beaches by foot is difficult. There is one permanent human settlement on Saona, Mano Juan, with a population of c. 300. Both Saona Island and Jaragua National Park are regularly visited by groups of itinerant fishermen for several days at a time.

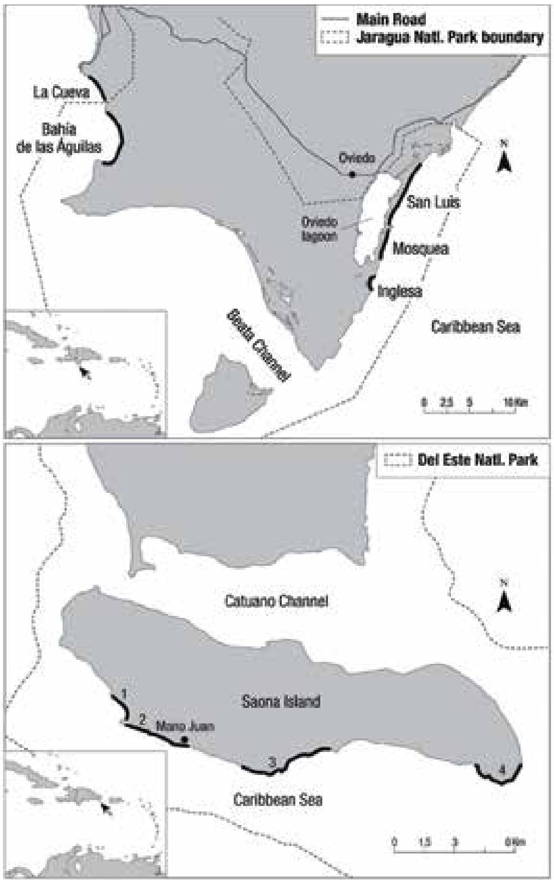

Figure 4.1. The two main study areas of (a) Jaragua National Park, indicating the beaches of La Cueva and Bahía de las Águilas in the west and of San Luis, Mosquea and Inglesa in the east, and (b) Saona Island, indicating Mano Juan village and the four surveyed areas of (1) El Toro, (2) Mano Juan, (3) Canto la playa and (4) Faro Punta Cana. The insets indicate the location of the main maps in the Caribbean.

Data collection

Daytime surveys were undertaken on foot by researchers and trained local people on the beaches of Jaragua National Park and Saona during 2006-2010. In Jaragua National Park beaches were monitored during: (1) 15 April to 19 November 2006, (2) 17 March to 20 October 2007, (3) 7 March to 8 October 2008, (4) 14 March to 23 September 2009, and (5) 26 March to 9 September 2010. Until 2009 the eastern beaches of the Park were monitored weekly, with the exception of the remote Inglesa beach, which was monitored twice per month. In 2009 and 2010 Mosquea was monitored daily by government rangers. On the western beaches of the Park there were 3-4 daytime surveys per week but survey effort ensured the record of all nesting events every year. Other remote beaches at this site with low levels of nesting were also occasionally visited to confirm reports of nests from reliable informants, although these were few (16 clutches in 5 years). On these beaches successful nesting events and false crawls that had occurred since the previous visit were recorded.

On Saona Island surveys started in September 2006 and thus we do not include total nesting numbers for this location and year. Since 23 September 2006 four major areas (Figure 4.1) were patrolled weekly throughout the year. Surveys in other areas of the Dominican Republic were undertaken during 2006-2010. Visits, sporadic patrols and interviews were made in 11 areas of the northern coast in 2006, 2007 and 2008, and on six beaches of the eastern and southern coasts in 2008, 2009 and 2010 (Figure 4.2). Reports about other nesting events, particularly from resorts or beach front developments, were also compiled.

During daytime surveys in Jaragua National Park and on Saona Island we recorded total number of emergences of nesting females. We identified the species based upon track characteristics (Schroeder and Murphy 1999) and confirmed whether each activity had successfully resulted in clutch deposition, by digging to find the clutch. Nest location was recorded using a global positioning system and turtle tracks were marked to avoid duplicate counts. Signs of eggs taken by people were recorded (presence of probing sticks, evidence of digging and broken eggshells, and/or human footprints around the nesting site; Troëng et al. 2004). Remaining clutches were either camouflaged and incubated under natural conditions on the beach, or clutches with high risk of subsequent take (because of proximity to a human settlement and the number of people in the area) were transferred to protected hatcheries for incubation in boxes. Because of funding limitations night surveys were only carried out on Saona Island and the western beaches of Jaragua National Park. To maximize the probability of finding nesting females during night surveys, we timed these surveys according to the inter-nesting intervals of the females. We collected data on clutch size and egg size and weight and measured the curved carapace length (CCL, from nuchal notch to the caudal tip of the carapace) and maximum curved carapace width (CCWmax) of nesting females. Leatherback females were tagged with large metal Inconel tags (National Brand and Tag Co., Newport, USA) fitted between the tail and rear flippers. Hawksbill females were tagged with small metal Inconel tags on the trailing edge of both fore-flippers.

RESULTS

Status and spatio-temporal distribution

The surveys confirmed that leatherback, hawksbill and green turtles are nesting in the Dominican Republic. Nesting is concentrated in Jaragua National Park and on Saona Island. Nesting outside these areas was confirmed, through visits and interviews, to be relatively uncommon (Figure 4.2).

The Jaragua National Park consistently had the highest number of clutches per year of leatherback turtles (mean 126.4 ± SD 74.1, range 17-210), with a total of 632 clutches recorded during 2006-2010 (Figures 4.2 and 4.3). These values correspond to an estimated annual number of three, 33, 19, 40 and 25 females in each of the 5 years, respectively (based on a mean number of clutches per season of 5.26; Boulon et al. 1996). Hawksbill turtles nested in low numbers in the Park (mean 14.6 ± SD 6.7 per year, range 7-22 clutches per year), with a total of 73 clutches recorded in the 5 years, mainly in the west (Figure 4.2). The estimated annual number of hawksbill turtles was five, four, four, one and two nesting females in each of the 5 years, respectively (based on a mean number of clutches per season of 4.5; Richardson et al. 1999). We recorded only one confirmed clutch of green turtles in this area during the study period (Figure 4.2).

Saona Island hosted the majority of recorded hawksbill (mean 100 ± SD 8.4 per year, range 93-111, total 400) and green turtle clutches (mean 9.2 ± SD 6.2, range 1-15, total 37) but only 22 leatherback turtle clutches were recorded (mean 5.5 ± SD 4.8, range 1-11). The annual number of hawksbill turtle clutches recorded was similar among years (Figure 4.3) and the estimated annual number of hawksbill females nesting in each year was 21, 23, 23 and 25 in 2007-2010, respectively.

Elsewhere 1-5 hawksbill turtle nests per year were reported on the Punta Cana and Cap Cana beach resort on the east coast. The only beach outside the two main nesting areas that seems to host significant leatherback turtle nesting is El Muerto beach (Figure 4.2). In 2009 we recorded 12 recent nests there. Local people reported nine nests later in the same season. This area merits future monitoring. The northern coast of Samaná Peninsula seems regularly to have a low number of nests (1-3 clutches per beach and year) of hawksbill and leatherback turtles, and merits further surveys.

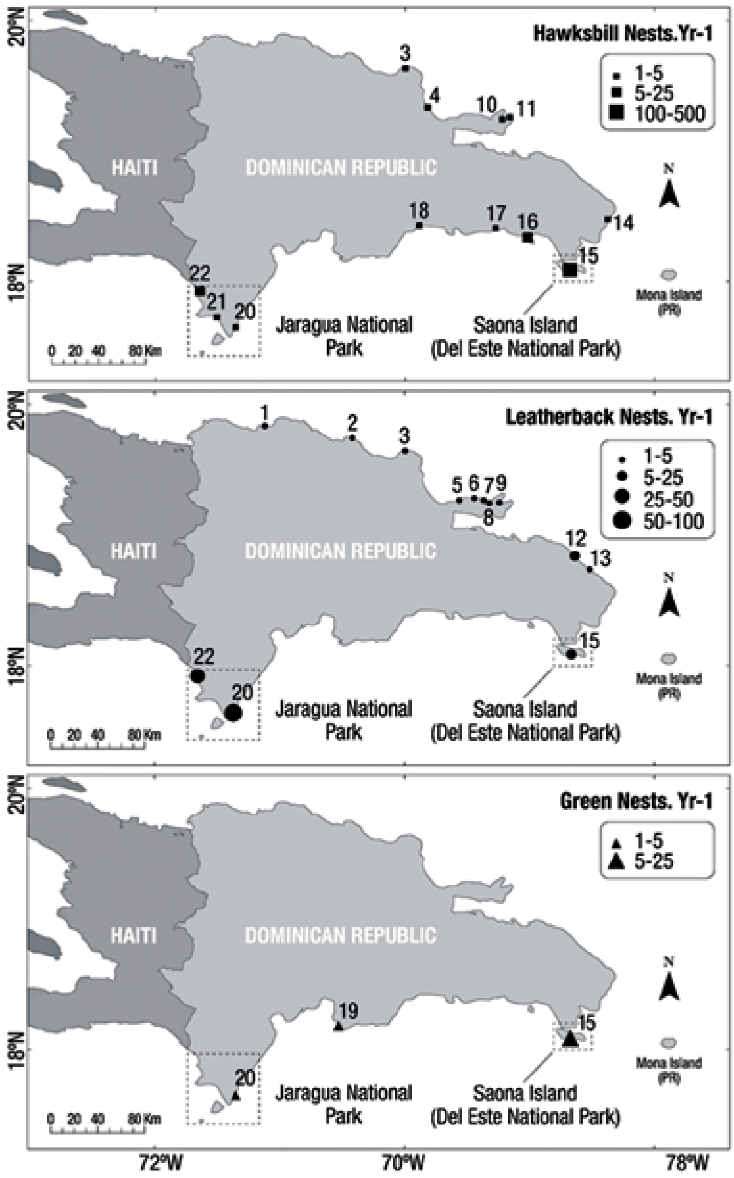

Figure 4.2. Maximum number of confirmed nests of hawksbill *Eretmochelys imbricata*, leatherback *Dermochelys coriacea* and green *Chelonia mydas* marine turtles per year during 2006-2010. Numbers correspond to beach locations: (1) beaches on both sides of the Estero Hondo bay (Punta Rucia and Punta Burén), (2) Sosúa and Cabarete beaches (Puerto Plata province), (3) Playa Grande, (4) Arroyo Salado, (5) Cosón (bonita and Morón beaches), (6) El Limón, (7) Lanza del Norte, (8) El Valle, (9) Rincón, (10) Colorá, (11) Las Galeras/Madama, (12) El Muerto/La Vacama, (13) Macao and Uvero Alto, (14) Punta Cana, (15) southern Saona Island (several beaches), (16) Isla Catalina, (17) San Pedro de Macorís, (18) Guibia (Santo Domingo), (19) Salinas, (20) Oviedo Lagoon, (21) Beata Channel, and (22) La Cueva-Bahía. Dotted lines show the two main study areas: Jaragua National Park in the south-west and Saona Island in the south-east (part of Del Este National Park).

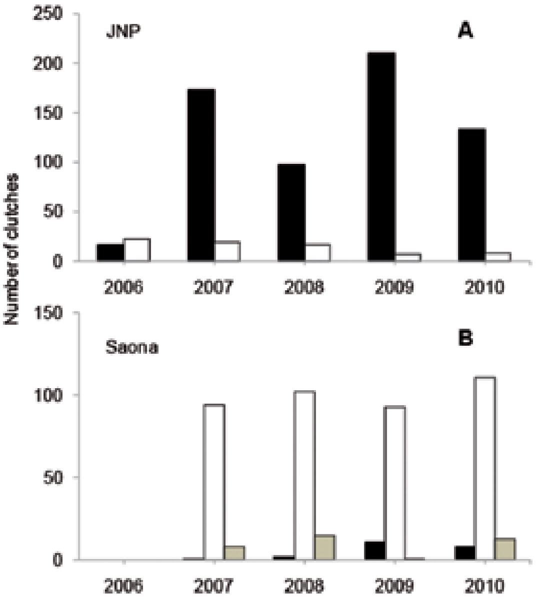

Figure 4.3. Number of clutches of leatherback (black), hawksbill (white) and green (grey) marine turtles per year in Jaragua National Park and on Saona Island. In 2007 one green turtle nest, which is not shown, was recorded in Jaragua National Park. Note the different y-axis scales.

Seasonality

In Jaragua National Park the nesting period of leatherback turtles extended from March to August, with most emergences occurring in April-June (89.6% of nests). The highest number of emergences was in May (mean number of nests 64 ± SD 12.7). Because of the low level of nesting by hawksbill turtles in the Park it is difficult to ascertain seasonality for this species there, although nesting appears to be higher in July (Figure 4.4). On Saona Island hawksbill turtles were observed nesting in every month of the year, although most nesting occurred in June-November (71.2% of the nests) and low levels of nesting were observed in December-May (Figure 4.4). The green turtle appears to have a shorter nesting season, from July to November, with a possible peak in August. However, any seasonality in the nesting of this species is unclear because of the low number of nests observed. The three species nested mainly at night although there were two records of hawksbill turtles nesting diurnally on Saona Island.

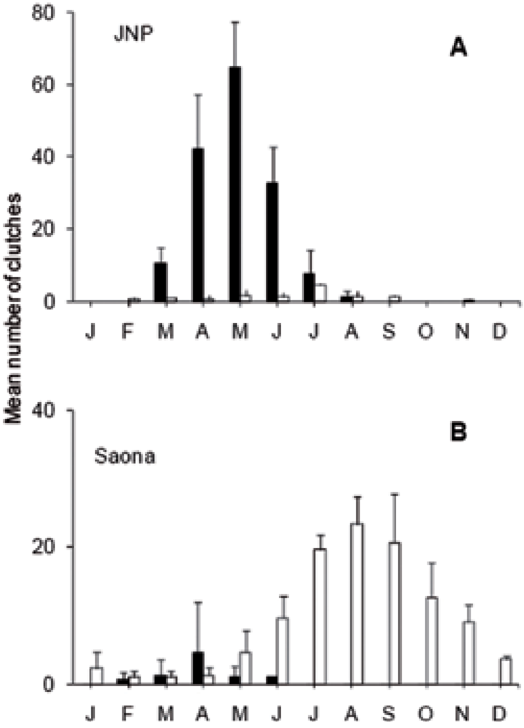

Figure 4.4. Mean (± SD) number of clutches per month of leatherback (black) and hawksbill (white) marine turtles for 2006-2010 in Jaragua National Park, and for 2007-2010 on Saona Island. Note the different y-axis scales.

Morphometrics and reproductive data

Biometric parameters of leatherback and hawksbill turtles are given in Table 4.1. All leatherback females encountered on western beaches of Jaragua National Park in 2006-2010 were tagged. We recorded four reproductive females (30.8%) with a CCL (132, 135, 138 and 143 cm) below the threshold carapace length for adult classification in leatherback turtles (145 cm; Eckert 2002; James et al. 2007). Mean clutch sizes (yolked eggs) of leatherback turtles recorded in the Park were 68.7 ± SD 18.1 (n = 64), 72.4 ± SD 15.5 (n = 59), 75.1 ± SD 17.3 (n = 125) and 67.5 ± SD 17.8 eggs (n = 67) in 2007-2010, respectively.

On Saona Island seven female hawksbill turtles were tagged in 2008, five in 2009 and seven in 2010 (Table 4.1). Mean clutch sizes of hawksbill turtles on Saona Island were 125.7 ± SD 23.3 (n = 29), 132.4 ± SD 29.9 (n = 55), 138.4 ± SD 23.2 (n = 40) and 139.5 ± SD 29 (n = 55) in 2007-2010, respectively.

Assessment of threats

Illegal taking of eggs was identified as the main threat to the marine turtles (Table 4.2). The highest level of egg-take was from the eastern beaches of Jaragua National Park. All clutches laid by hawksbill and leatherback turtles on Inglesa beach were taken by humans. Overall 88.9% of hawksbill clutches laid on Mosquea and San Luis beaches during the study period were taken and only a small percentage of leatherback clutches were incubated on the beach (1% in 2007 and 8.4% in 2009); the remainder were relocated to a protected hatchery for artificial incubation or otherwise taken by humans. The percentage of leatherback turtle clutches taken on the beaches of Bahía de las Águilas and La Cueva in the west of the Park was lower than on the eastern beaches, with 75.9% of 290 clutches incubated naturally during the study period. However, take of hawksbill turtle clutches was high every year (Table 4.2).

We found hawksbill and green turtle carcasses, bones and scutes from both adults and juveniles in Jaragua National Park and on Saona Island, which we presume to be remains, at least in part, from illegal take. Meat and eggs are consumed and illegally sold in local markets (YML and JT, pers. obs.). Throughout the study period no reported cases of consumption of leatherback meat were recorded.

Table 4.1. Morphometric and reproductive parameters of leatherback (*Dermochelys coriacea*) and hawksbill (*Eretmochelys imbricata*) turtles nesting at Jaragua National Park and Saona Island in the Dominican Republic (Figure 4.1).

	Leatherback (JNP)			Hawksbill (Saona)		
	N	Mean ± SD	Range	N	Mean ± SD	Range
Curved carapace length (cm)	13	147.4 ± 8.7	132-162	19	87.2 ± 4.7	81-96
Curved carapace width (cm)	13	107.7 ± 5.1	100-118	15	77.1 ± 4.2	71-84
Clutch size (no. of yolked eggs)	315	71.7 ± 17.5	10-128	179	134.8 ± 27.5	82-235
Egg size (mm)	740	5.2 ± 0.2	4.5-5.7	380	3.4 ± 0.09	3.1-3.8
Egg mass (g)	500	85.4 ± 8.7	65.5-102.4	240	29.9 ± 2.3	24.5-85.4

Table 4.2. Total numbers and percentage of hawksbill, leatherback and green (*Chelonia mydas*) turtle clutches recorded in Jaragua National Park and Saona Island during 2006-2010 that were subject to human take, translocated to a breeding facility, or incubated on the beach under natural conditions.

	Leatherback				Hawksbill				Green			
Year (by location)	Total	Human take (%)	Translocated (%)	Beach (%)	Total	Human take (%)	Translocated (%)	Beach (%)	Total	Human take (%)	Translocated (%)	Beach (%)
JARAGUA NATIONAL PARK (WEST)												
2006	2	50.0	0.0	50.0	22	68.1	27.2	4.5				
2007	59	37.3	10.2	52.5	14	71.4	28.6	0.0	1	0.0	100.0	0
2008	63	1.6	34.9	63.5	15	33.3	66.7	0.0				
2009	83	4.8	16.9	78.3	4	25.0	50.0	25.0				
2010	83	0.0	0.0	100.0	8	37.5	12.5	50.0				
JARAGUA NATIONAL PARK (EAST)												
Mosquea-San Luis												
2006	7	42.8	57.1	0.0								
2007	102	48.0	51.0	1.0	4	75.0	0.0	25.0				
2008	29	44.8	55.2	0.0	2	100.0	0.0	0.0				
2009	119	23.5	68.1	8.4	3	100.0	0.0	0.0				
2010	39	46.1	51.3	2.6								
Inglesa												
2006	8	100.0	0.0	0.0	1	100.0	0.0	0.0				
2007	12	100.0	0.0	0.0								
2008	6	100.0	0.0	0.0								
2009	8	100.0	0.0	0.0								
2010	12	100.0	0.0	0.0								
SAONA												
2007	1	100.0	0.0	0.0	94	56.4	21.3	22.3	8	37.5	37.5	25.0
2008	2	50.0	50.0	0.0	102	32.3	50.0	17.6	15	13.3	73.3	13.3
2009	11	72.7	9.1	18.2	93	50.5	41.9	7.5	1	0.0	100.0	0.0
2010	8	75.0	25.0	0.0	111	17.2	46.8	36.0	13	7.7	38.5	53.8

DISCUSSION

For the first time in recent decades we have documented the presence of three marine turtle species (hawksbill, leatherback and green) nesting in the Dominican Republic. Comparison of our results with early reports (Ottenwalder 1981; Ross and Ottenwalder 1983) indicates that a profound decline appears to have taken place in the last 30 years. Nesting has largely been reduced to the undeveloped, protected areas of Jaragua National Park and Saona Island. However, there are still high rates of human exploitation of eggs and at least some take of turtles in these protected areas.

The number of clutches of hawksbill and green turtles, which we found mainly on Saona Island, are low compared to estimates of 400 hawksbill and 260 green turtles nesting per year throughout the country in the 1980s (Ottenwalder 1981). Ross and Ottenwalder (1983) suggested that 300 leatherback turtles were nesting annually in the Dominican Republic, based on one aerial survey, 24 interviews and a few beach surveys between 24 March and 13 April 1980. Based on 5 years of systematic surveys we have estimated that a maximum of 40 leatherback females currently nest in Jaragua National Park per year, with no more than 50 in the country. Of further concern was our failure to detect loggerhead turtles nesting, even though Ottenwalder (1981) estimated c. 60 females were nesting per year.

Our findings seem to indicate that there has been a marked reduction in the abundance of the four species throughout the country since the 1980s. However, there are some caveats regarding the previous estimates. Firstly, data reported by Ottenwalder (1981) were based on 2 years of non-intensive monitoring and interviews. Secondly, loggerhead turtles are considered to nest infrequently in the Caribbean (Ehrhart et al. 2003; Dow et al. 2007), and Ottenwalder noted that, in interviews, many fishermen confused this species with others. It is therefore possible that the number of loggerhead turtles nesting was previously overestimated, as is thought to have occurred in similar studies based on interview data in the region (Richardson et al. 2009). Thirdly, the decline of the leatherback turtle nesting rookeries in the Dominican Republic could be because of a shift of nesting to nearby rookeries, as this species has low nest site fidelity (Georges et al. 2007; Troëng et al. 2007). The leatherback turtle population in the Dominican Republic possibly forms part of a wider regional nesting stock, as has been suggested for other rookeries in the Antilles (Dutton et al. 2005). Despite these caveats it is probable that harvesting of eggs and females from beaches and the trade in marine turtles documented in the Dominican Republic (Ottenwalder 1996; Fleming 2001; Reuter and Allan 2006) has contributed significantly to the decline of these turtle populations. However, we cannot refute the possibility that numbers of all three turtle species nesting in the Dominican Republic, particularly the leatherback and hawksbill, were lower in previous years and are beginning, as in some Caribbean range states, to increase (Dutton et al. 2005; Troëng and Rankin 2005; Richardson et al. 2006).

According to Ottenwalder (1981) hawksbill turtles nested in the past in significant numbers on beaches around the entire coast of the Dominican Republic and green turtles nested every night on northern beaches. Currently, only the relatively unspoilt beaches of Saona Island, in Del Este National Park, provide major nesting areas for hawksbill and green turtles. Nesting of leatherback turtles previously occurred on the northern and eastern coasts, and the beaches of Macao and El Muerto in La Altagracia province on the eastern coast hosted 2-3 females nightly during the nesting season in the 1980s (Ross and Ottenwalder 1983). Significant nesting by leatherback turtles occurs only on the isolated beaches of Jaragua National Park, where access for people is difficult and there is no coastal development. In addition to the taking of eggs and adults on beaches near habitation, habitat has been lost to intensive coastal development during the last 30 years (Ottenwalder 1981; León 2004; Wielgus et al. 2010). Currently, the northern and eastern coasts have the most tourism infrastructure and we detected only sporadic nesting events on these beaches.

Despite the importance of coastal and marine protected areas as refuges for marine biodiversity in the country we observed that there is little actual protection and inadequate management by local authorities (OR, YML, PF and JT, pers. obs.). Our results highlight the need for adequate protection and management of these areas for marine turtle conservation, especially considering that Jaragua National Park also hosts important foraging habitats for Caribbean hawksbill and green turtles (León and Diez 1999; León and Bjorndal 2002).

Items made from tortoiseshell were previously widely available in tourist gift shops in the Dominican Republic (Ottenwalder 1996; Mota and León 2003; Reuter and Allan 2006; Feliz et al. 2010). However, since 2009 the environmental authorities have conducted several seizures of tortoiseshell items in the gift shops of Santo Domingo, resulting in a significant reduction in the sale of such items (OR, YML and PF, pers. obs.).

Our study has shown that the leatherback turtle nesting season is from March to August, with a peak in May, consistent with reports for neighbouring nesting populations (Dutton et al. 2005; Hilterman and Goverse 2007; McGowan et al. 2008). The seasonality of hawksbill turtle nesting is similar to nearby rookeries on Antigua, Barbados and Mona Island, Puerto Rico (Richardson et al. 1999; Beggs et al. 2007; Diez and van Dam 2007).

The differences in nesting activity between the two main nesting areas may be explained by species' preferences. Leatherback turtles tend to use high energy beaches with deep water nearby, free of obstructions and often with a windward orientation (Mrosovsky 1983; Eckert 1987). This description fits the eastern beaches of Jaragua National Park, which is the most important nesting area for the species in the country. Hawksbill turtles, however, often swim over reefs and through shallow waters with lower wave energy to reach beaches with dense vegetation (Horrocks and Scott 1991). This description fits all of the Saona Island beaches and most of the western beaches of Jaragua National Park.

The mean CCL (147.4 cm) of leatherback turtles in the Dominican Republic is lower than the global mean, which is normally distributed around 155 cm (see review in

Stewart et al. 2007) and we confirmed the viability of clutches of females that are below the minimum size of sexual maturity reported elsewhere (e.g. Eckert 2002). We are confident that our results are not measurement errors because we used standard measurements (Bolten 1999) and the carapaces had no deformities or injuries. This could be the sign of a recovering population with a high proportion of neophyte females. However, reduction in the size of nesting females can be caused by intensive capture of turtles either in the foraging areas or in the same nesting areas (Bjorndal et al. 1985; Limpus et al. 2003). The mean CCL of nesting hawksbill turtles in the Dominican Republic is similar to the mean CCL in other Caribbean regions. However, the CCL of 11 hawksbill females measured on Saona Island was within the CCL range recorded for neophytes in Buck Island, US Virgin Islands (82.1-88.0 cm, Garland and Hillis-Starr 2003).

This research has filled a significant gap in knowledge of the population status and conservation of marine turtles in the Caribbean (Dow et al. 2007). Many of the large turtle rookeries in the region are the subject of ongoing long-term monitoring and conservation programmes. Although several appear to be showing signs of recovery (St Croix, US Virgin Islands, Dutton et al. 2005; Barbados, Beggs et al. 2007; Mona Island in Puerto Rico, Diez and van Dam 2007) this is not the case for all turtle rookeries in the region.

To facilitate effective marine turtle conservation across the Caribbean conservation programmes need to focus on sites that were formerly important nesting areas (McClenachan et al. 2006). Our results suggest this is the case in the Dominican Republic, which therefore warrants increased attention at the regional level, particularly for the hawksbill turtle. Molecular and genetic techniques have provided insights into patterns of migration, stock differentiations, and links between nesting beaches and foraging grounds of the hawksbill turtle in the Caribbean (Bowen et al. 2007; Blumenthal et al. 2009). Recovery will therefore depend on actions both in the Dominican Republic and in the wider Caribbean.

The fact that nesting activity is concentrated in protected areas provides an opportunity to implement specific actions for protection and management of the rookeries (Santidrián Tomillo et al. 2008). We conclude that to improve marine turtle conservation in the Dominican Republic it is necessary to promote the effective management of existing protected areas, including nesting beaches for marine turtles, to promote the management of coastal development to avoid further loss of nesting beaches, and to support the enforcement and efficient implementation of laws to reduce take of marine turtles and their eggs.

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REFERENCES

Beggs JA, Horrocks JA and Krueger BH (2007) Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies. Endangered Species Research, 3, 159 - 168.

Bell CD, Blumenthal JM, Austin TJ, Solomon JL, Ebanks-petrie G, Broderick AC and Godley BJ (2006) Traditional Caymanian fishery may impede local marine turtle population recovery. Endangered Species Research, 2, 63 - 69.

Bjorndal KA, Carr A, Meylan AB and Mortimer JA (1985) Reproductive biology of the hawksbill turtle *Eretmochelys imbricata* at Tortuguero, Costa Rica, with notes on the ecology of the species in the Caribbean. Biological Conservation, 34, 353 - 368.

Bjorndal KA and Jackson JBC (2003) Roles of sea turtles in marine ecosystems: reconstructing the past. In: Lutz PL, Musick JA and Wyneken J (eds). The Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, USA, pp 259 - 273.

Blumenthal JM, Abreu-Grobois FA, Austin TJ, Broderick AC, Bruford MW, Coyne MS Ebanks-Petrie G, Formia A, Meylan PA, Meylan AB and Godley BJ (2009) Turtle groups or turtle soup: dispersal patterns of hawksbill turtles in the Caribbean. Molecular Ecology, 18, 4841 - 4853.

Bolten AB (1999) Techniques for measurement of sea turtles. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donelly M (eds). Research and Management Techniques for the Conservation of Sea Turtles. IUCN/Species Survival Commission Marine Turtle Specialist Group Publication No. 4. Blanchard, USA, pp 110 - 114.

Boulon RH, Dutton PH and McDonald DL (1996) Leatherback turtles (*Dermochelys coriacea*) on St. Croix, US Virgin Islands: fifteen years of conservation. Chelonian Conservation and Biology, 2, 141 - 147.

Bowen BW, Grants WS, Hillis-Star Z, Shaver DJ, Bjorndal KA, Bolten AB and Bass AL (2007) Mixed stock analysis reveals the migrations of juvenile hawksbill, *Eretmochelys imbricata*, in the Caribbean Sea. Molecular Ecology, 16, 49 - 60.

Bräutigam A and Eckert KL (2006) Turning the Tide: Exploitation, Trade and Management of Marine Turtles in the Lesser Antilles, Central America, Colombia and Venezuela. TRAFFIC nternational, Cambridge, UK.

Diez CE and van Dam RP (2007) Mona and Monito Island—Puerto Rico Hawksbill Turtle Research Project: Research Report for 2006. Unpublished Technical Report. Mona Island Hawksbill Research Group in collaboration with Departamento de Recursos Naturales y Ambientales de Puerto Rico, Chelonia, Inc., and Universidad de Puerto Rico, Rio Piedras, Puerto Rico.

Dominguez TJ and Villalba AA (1994) Trade of hawksbill carapaces in Santo Domingo, Dominican Republic. In: Bjorndal KA, Bolten AB, Johnson DA and Eliazar PJ (eds). Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. Hilton Head, South Carolina, USA, pp 34 - 35.

Dominici GO (1996) Monitoreo de anidamiento de tortuga tinglar *Dermochelys coriacea* en playas del Parque Nacional Jaragua. In: Memorias del II congreso de la Biodiversidad Caribeña, Santo Domingo, Dominican Republic.

Dow W, Eckert K, Palmer M and Kramer P (2007) An Atlas of Sea Turtle Nesting Habitat for the Wider Caribbean Region. WIDECAST Technical Report No. 6. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, Beauford, USA.

Dutton DL, Dutton PH, Chaloupka M and Boulon RH (2005) Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biological Conservation, 126, 186 - 194.

Eckert KL (1987) Environmental unpredictability and leatherback sea turtle (*Dermochelys coriacea*) nest loss. Herpetologica, 43, 315 - 323.

Eckert SA (2002) Distribution of juvenile leatherback sea turtle *Dermochelys coriacea* sightings. Marine Ecology Progress Series, 230, 289 - 293.

Ehrhart LM, Bagley DA and Redfoot WE (2003) Loggerhead turtles in the Atlantic Ocean: geographic distribution, abundance, and population status. In: Bolten AB and Witherington BE (eds). Loggerhead Sea Turtles. Smithsonian Institution, Washington, DC, USA, pp 157 - 174.

Feliz P, León YM, Tomás J, Hierro K, Mateo A, Méndez MD and Raga JA (2010) Tortoiseshell trade in Santo Domingo, Dominican Republic: discouraging news for Caribbean hawksbill. In: Dean K and López Castro MC (eds). Proceedings of the Twenty-Eighth Annual Workshop on Sea Turtle Biology and Conservation. National Oceanic and Atmospheric Administration, Miami, USA, pp 247.

Fleming EH (2001) Swimming against the tide: recent surveys of exploitation, trade, and management of marine turtles in the Northern Caribbean. TRAFFIC North America, Washington, DC, USA.

Garland K and Hillis-Starr Z (2003) Carapace characteristics of hawksbills (*Eretmochelys imbricata*) at Buck Island National Monument, US Virgin Islands: long term remigrants vs neophytes. In: Seminoff JA (ed). Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. National Oceanic and Atmospheric Administration, Miami, USA, pp 286.

Georges JY, Fossette S, Billes A, Ferraroli S, Fretey J, Grémillet D, Le Maho Y, Myers AE, Tanaka H and Hays GC (2007) Meta-analysis of movements in Atlantic leatherback turtles during the nesting season: conservation implications. Marine Ecology Progress Series, 338, 225 - 232.

Hilterman M and Goverse E (2007) Nesting and nest success of the leatherback turtle (*Dermochelys coriacea*) in Suriname, 1999-2005. Chelonian Conservation and Biology, 6, 87 - 100.

Horrocks JA and Scott NM (1991) Nest site location and nest success in the hawksbill turtle *Eretmochelys imbricata* in Barbados, West Indies. Marine Ecology Progress Series, 69, 1 - 8.

James MC, Sherrill-Mix SA and Myers RA (2007) Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series, 337, 245 - 254.

León YM (2004) Community Impacts of Coastal Tourism in the Dominican Republic. Ph.D. Dissertation. University of Rhode Island, Kingston, Rhode Island, USA.

León YM and Bjorndal KA (2002) Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. Marine Ecology Progress Series, 245, 249 - 258.

León YM and Diez CE (1999) Population structure of hawksbill turtles on a foraging ground in the Dominican Republic. Chelonian Conservation and Biology, 3, 230 - 236.

Limpus CJ, Miller J, Parmenter C and Limpus D (2003) The green turtle, *Chelonia mydas*, population of Raine Island and the northern Great Barrier Reef 1843-2001. Memoirs of the Queensland Museum, 49, 349 - 440.

McClenachan L, Jackson J and Newman M (2006) Conservation implications of historic sea turtle nesting beach loss. Frontiers in Ecology and the Environment, 4, 290 - 296.

McGowan A, Broderick AC, Frett G, Gore S, Hastings M, Pickering A, Wheatley D, White J, Witt MJ and Godley BJ (2008) Down but not out: marine turtles of the British Virgin Islands. Animal Conservation, 11, 92 - 103.

Mota JM and León YM (2003) Beliefs and perceptions associated with sea turtle products in the Dominican Republic. In: Pilcher NJ (ed). Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. Kuala Lumpur, Malaysia, pp 197 - 199.

Mrosovsky N (1983) Ecology and nest-site selection of leatherback turtles, *Dermochelys coriacea*. Biological Conservation, 26, 47 - 56.

Ottenwalder JA (1981) Estudio preliminar sobre el estado, distribución, y biología reproductiva de las tortugas marinas en la República Dominicana. Tesis de Licenciatura, Universidad Autónoma de Santo Domingo, Dominican Republic.

Ottenwalder JA (1987) National Report for Dominican Republic. Western Atlantic ymposium II, Mayagüez, Puerto Rico, September 1987. WATS2 - 072. Unpublished.

Ottenwalder JA (1996) Conservation and Management of Sea Turtles in the Dominican Republic. Unpublished Report.

Parsons JJ (1962) The Green Turtle and Man. University of Florida Press, Gainesville, USA.

Reuter A and Allan C (2006) Tourists, Turtles and Trinkets: A Look at the Trade in Marine Turtle Products in the Dominican Republic and Colombia. TRAFFIC North America, Washington, DC, USA.

Richardson JI, Bell R and Richardson TH (1999) Population ecology and demographic implications drawn from an 11-year study of nesting hawksbill turtles, *Eretmochelys imbricata*, at Jumby Bay, Long Island, Antigua, West Indies. Chelonian Conservation and Biology, 3, 244 - 250.

Richardson JI, Hall DB, Mason PA, Andrews KM, Bjorkland R, Cai Y and Bell R (2006) Eighteen years of saturation tagging data reveal a significant increase in nesting hawksbill sea turtles (*Eretmochelys imbricata*) on Long Island, Antigua. Animal Conservation, 9, 302 - 307.

Richardson PB, Bruford M, Calosso MC, Campbell LM, Clerveau W, Formia A, Godley BJ, Henderson C, McClellan K, Newman S, Parsons K, Pepper M, Ranger S, Silver JJ, Slade L and Broderick AC (2009) Marine turtles in the Turks and Caicos Islands: remnant rookeries, regionally significant foraging stocks and a major turtle fishery. Chelonian Conservation and Biology, 8, 192 - 207.

Ross JP and Ottenwalder JA (1983) The leatherback sea turtle, *Dermochelys coriacea*, nesting in the Dominican Republic. In: Rhodin AGJ and Miyata K (eds). Advances in Herpetology and Evolutionary Biology. Museum of Comparative Zoology, Harvard University, Cambridge, USA, pp 706 - 712.

Santidrián Tomillo P, Saba VS, Piedra R, Paladino FV and Spotila JR (2008) Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. Conservation Biology, 22, 1216 - 1224.

Schroeder B and Murphy S (1999) Population surveys (ground and aerial) on nesting beaches. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donelly M (eds). Research and Management Techniques for the Conservation of Sea Turtles. IUCN/Species Survival Commission Marine Turtle Specialist Group Publication No. 4. Blanchard, USA, pp 45 - 53.

Stam S and Stam R (1992) Turtle trouble in the Dominican Republic. Marine Turtle Newsletter, 57, 19 - 21.

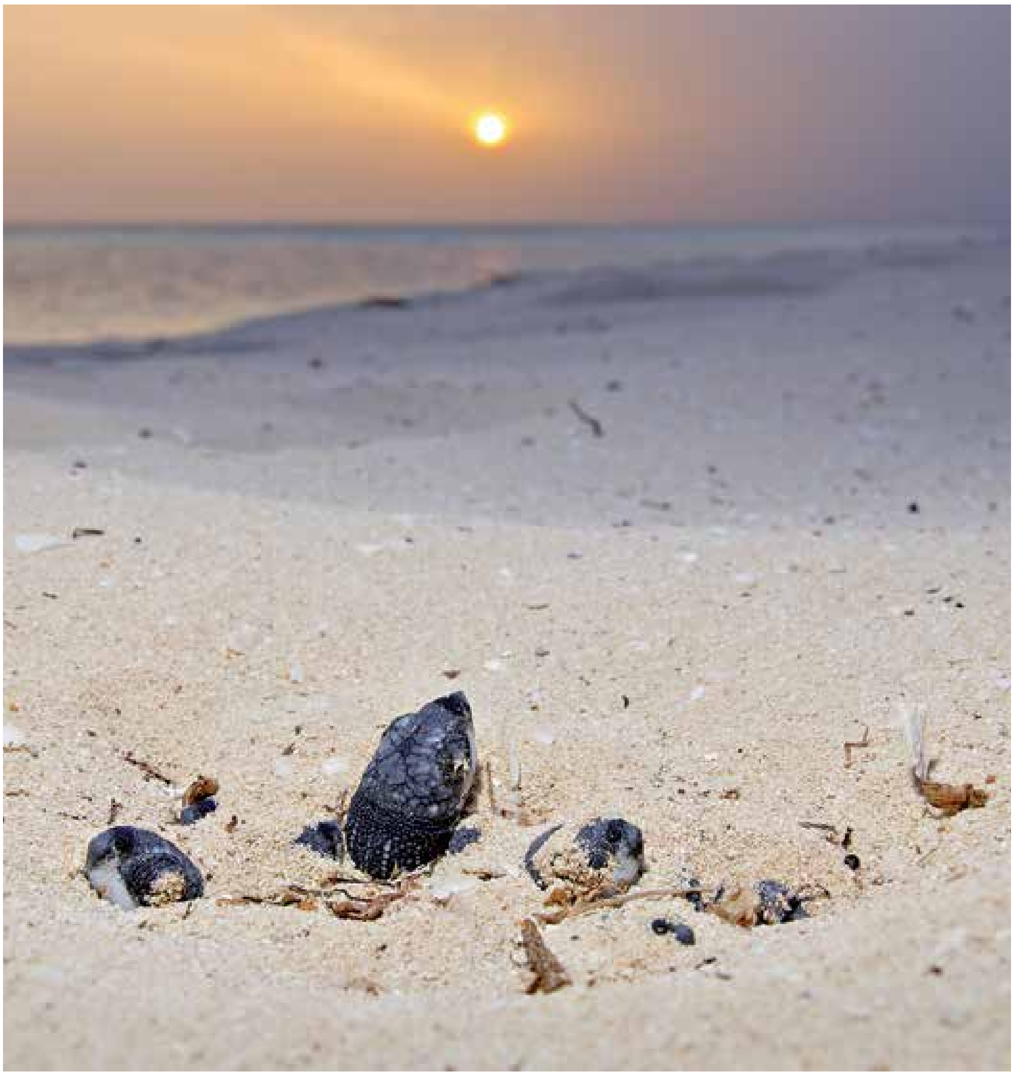
Stewart K, Johnson C and Godfrey MH (2007) The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean Basins. Herpetological Journal, 17, 123 - 128.

Troëng S, Chacón D and Dick B (2004) Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. Oryx, 38, 395 - 403.

Troëng S, Harrison E, Evans D, De Haro A and Vargas E (2007) Leatherback turtle nesting trends and threats at Tortuguero, Costa Rica. Chelonian Conservation and Biology, 6, 117 - 122.

Troëng S and Rankin E (2005) Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biological Conservation, 121, 111 - 116.

Wielgus J, Cooper E, Torres R and Burke L (2010) Coastal Capital: Dominican Republic. Case Studies on the Economic Value of Coastal Ecosystems in the Dominican Republic. Working paper. World Resources Institute, Washington, DC, USA.



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CHAPTER V:

ASSESSING THE EFFICACY OF DIRECT CONSERVATION INTERVENTIONS: CLUTCH PROTECTION OF THE LEATHERBACK MARINE TURTLE IN THE DOMINICAN REPUBLIC

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ASSESSING THE EFFICACY OF DIRECT CONSERVATION INTERVENTIONS: CLUTCH PROTECTION OF THE LEATHERBACK MARINE TURTLE IN THE DOMINICAN REPUBLIC

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ABSTRACT

In the Dominican Republic (DR), northern Caribbean, the beaches of the Jaragua National Park (JNP, southwest of DR) are the last major nesting site for the leatherback marine turtle, *Dermochelys coriacea*, in the country (126.4 ± SD 74.1, clutches per year, range (17-210): 2006-2010). This nesting aggregation is highly threatened due to a widespread illegal egg take. Clutch relocation and artificial incubation have been carried out here as a protection measure since 1974. We sought to critically assess the efficacy of such efforts and investigated how artificial incubation may be influencing success of clutches and resultant sex ratios. We compared hatching success, incubation duration and embryo mortality from *in situ* clutches (n = 43 clutches) with those artificially incubated at sites in the east and west of the park (n = 35 and n = 31 clutches, respectively). Our study found that in the west artificial incubation significantly decreased hatching success in clutches and, in the east, increased incubation duration, which we predict would result in an increase in male production from these clutches. Clutch relocation is currently the only viable conservation option for clutches on eastern beaches due to intense egg take, but steps are needed to ensure that natural sex ratio is not distorted. However, on the western beaches, *in situ* clutch incubation seems possible through beach protection. Further community engagement and enforcement is required to improve conservation measures at eastern beaches if long-term, less sustainable, intervention is to be avoided.

INTRODUCTION

In critical conservation situations including endangered species urgent actions are demanded to preserve species and ecosystems (Andrews 2000). However, in many situations conservation actions have been passed on from individual to individual with no assessment to see whether or not these management actions fulfill the conservation objectives (Pullin and Knight 2003). Carrying out evaluations of these activities is necessary to demonstrate that conservation actions actually achieve their objectives (Pullin and Knight 2009; Sutherland et al. 2009).

As a result of centuries of exploitation marine turtles are recognised internationally as species of conservation concern (Hamman et al. 2010). The take of eggs by coastal people, once widespread around the world, is still a significant threat to the survival of some marine turtle populations (Santidrián Tomillo et al. 2008). This has led to a variety of conservation measures, including protection of nesting beaches (Frazier 2002; Santidrián Tomillo et al. 2009), and programs of egg relocation from threatened sites to other beach locations or to enclosed hatcheries (Tuttle and Rostal 2010; Liles et al. 2011).

On nesting beaches where clutches are handled, management techniques carried out may affect offspring production (Pintus et al. 2009). The handling and transportation of eggs may result in embryo death as rotation or vibration can rupture the embryonic membrane (Phillott and Parmenter 2007), resulting in a reduced hatching success of relocated clutches when compared to those incubated *in situ* (Eckert and Eckert 1990; Özdemir and Türkozan 2006). Temperature, humidity and oxygen levels in relocated clutches are likely to be different from those chosen by the nesting female, therefore hatching success, sex ratios and phenotype may be affected (Foley et al. 2000). In marine turtles, the temperature experienced by the embryo during the middle third of development determines the sex of the hatchling (Yntema and Mrosovsky 1982) and while female biased offspring sex ratios appear to be the norm for marine turtles, there have been examples of artificial incubation carried out under cooler thermal conditions resulting in male biased offspring sex ratios (Mrosovsky 1982; Morreale et al. 1982; Dutton et al. 1985). Furthermore, temperature may also affect the embryonic development with very high or very low incubation temperatures increasing embryo mortality rates (Ackerman 1997; Broderick et al. 2001).

The leatherback marine turtle, *Dermochelys coriacea*, is currently listed as critically endangered by the world conservation union (IUCN 2012). However, the population status of this species varies among locations. Leatherback nesting populations in the Indo-Pacific have declined precipitously in recent decades (Spotila et al. 2000; Sarti Martínez et al. 2007) whilst many Atlantic nesting populations are stable or increasing (Dutton et al. 2005; McGowan et al. 2008; Witt et al. 2011). The Wider Caribbean region holds some of the globally important nesting sites, such as Trinidad and Tobago (Dow and Eckert 2011) and the Caribbean coast of Colombia and Panama (Patiño-Martinez et al. 2008). However, there are many small, widely dispersed sites that may play an important role in species recovery that lack intensive population monitoring effort (Dow et al. 2007).

In the Dominican Republic, the leatherback turtle nests sporadically around the coast, but the beaches of the Jaragua National Park (JNP, southwest of DR) have been recently identified as the last important stable nesting site for the species in the country, (mean of 126.4 clutch.year^{-1} SD 74.1, range 17-210 over 5 seasons; 2006 to 2010; Revuelta et al. 2012). Recent analyses with genetic markers show that leatherback turtles in the Dominican Republic appear to be connected with many other Caribbean populations (Carreras et al. 2013). Although the harvesting of turtles or their products has been banned since 1966, the enforcement of environmental law in the DR is weak and the leatherback nesting stock of JNP is highly threatened due to a widespread illegal egg take (Revuelta et al. 2012). However, there is marked spatial heterogeneity in the level of human predation; illegal egg take is close to 100% in the eastern beaches, whilst egg take is much lower at western beaches (75.9% of clutches could be incubated *in situ* 2006 to 2010; Revuelta et al. 2012). In the face of this pressure there is a 38 year history of artificial incubation in JNP. In 1974, a local assistant was trained to carry out relocation and artificial incubation of eggs in Styrofoam boxes (Ottenwalder 1981), placing them in a facility located 10 km from the nesting beach. However, this activity has been carried out sporadically with no strict protocol or scientific monitoring. In 2006 we initiated a project to assess this conservation program investigating how artificial incubation may be influencing hatching success of clutches and resultant sex ratios.

MATERIALS AND METHODS

Study Site

Jaragua National Park is a protected area of 1374 km^2 (of which 905 km^2 is marine reserve) situated in the south western corner of the DR (Figure 5.1). There are five main turtle nesting beaches grouped in two areas: (1) Bahía de las Águilas (4.4 km) and La Cueva beach (2.5 km) (western beaches [WB], N17° 57', W71° 39'), and (2) Mosquea beach (3.3 km), San Luis beach (11 km) and Inglesa beach (1.2 km) (eastern beaches [EB], N17°44', W71° 20') (Figure 5.1). The two areas are separated by approximately 50 km. The western beaches border calm shallow waters and have fine-grained, white coralline sand (width ranges: 4-20 m) backed by low scrub vegetation. A small village (around 20 inhabitants) is located between the two beaches, which, in addition, are frequented by several thousand tourists per annum.

The eastern beaches (EB) lay beyond the Oviedo Lagoon, and are generally wider than the western ones (width ranges: 7-40 m), and have coarser, yellow sand, and steeper beach slope. They are exposed to the prevailing north-easterly winds and currents which deposit large amounts of plastic and other debris. Although not frequented by tourists, these eastern beaches are patrolled by beachcombers, some of whom partake in a high level of illegal egg take for consumption and sale (Revuelta et al. 2012).

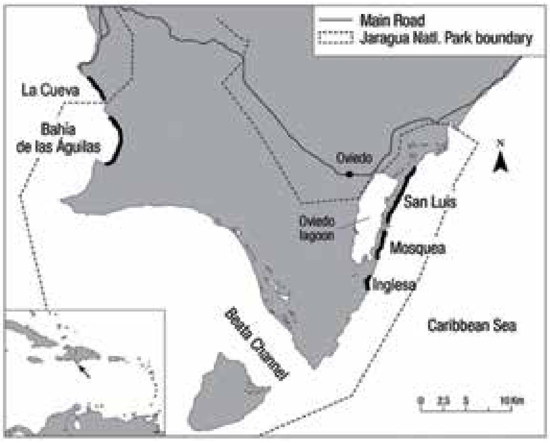

Figure 5.1. Map of JNP, indicating the beaches of La Cueva and Bahía de las Águilas (western beaches) and of San Luis, Mosquea and Inglesa (eastern beaches).

Surveys

In 2006 and 2007, we carried out preliminary weekly beach surveys on the western and eastern beaches of JNP to obtain baseline data on marine turtle nesting activity. Based on the observed differences in human egg take between western and eastern beaches and the availability of human resources, we applied different survey and conservation strategies in each area of study in subsequent years. Although some artificial incubation was carried out in these two seasons, it was not until 2008 that we were able to test robustly for effects of artificial incubation interventions.

Eastern beaches: In 2008 we carried out daytime weekly surveys during the leatherback nesting season (from March until August, Revuelta et al. 2012) with the exception of the remote Inglesa beach which was monitored twice a month. In 2009, added to our weekly surveys, governmental rangers performed nightly beach surveys in Mosquea and San Luis beaches and weekly surveys in Inglesa beach. Owing to the high level of egg take at these beaches the strategy chosen was the relocation and artificial incubation of all clutches that were not-predated at the time of recording.

Western beaches: In 2008 and 2009, researchers and governmental rangers carried out daily surveys from March to August. Night patrols were also carried out two or three nights per week during the busiest part of the laying season (April and May, Revuelta et al. 2012) in order to witness laying events. These beaches are rarely visited by individuals involved in illegal activities and access is controlled by rangers. For this reason, we left 50% (n = 22 clutches) and 76.9% (n = 40 clutches) of clutches incubating *in situ* in 2008 and 2009, respectively. In these years we limited relocation to clutches that were thought unlikely to hatch without intervention (i.e., those clutches laid in a sand road near the beach or clutches threatened by tidal inundation). From 2010 onwards, we left 100% of clutches incubating *in situ*.

From 2008 we relocated clutches for artificial incubation into the park rangers' barracks. The facilities were a similar size and had the same characteristics at both areas: wooden barracks with concrete floor and roof made from corrugated metal panels. The WB barracks is on a hilltop about 2 km away from the beaches, and it is only accessible via a dirt track. This facility is located in an area of arid scrub and is exposed to the sun all day and to dry winds. Precipitation is more frequent at EB and the rangers' barracks lies directly behind the Mosquea beach, inside a forest and near the Oviedo lagoon (See Figure 5.1).

Egg relocation and incubation

During daytime surveys, clutches were located by careful probing of the sand with a stick at nesting sites discovered from tracks (Schroeder and Murphy 1999). During night surveys eggs were collected after the female had returned to the sea.

For artificial incubation we carefully excavated the egg chamber by hand and transferred the yolked eggs into polyethylene exterior boxes with polyurethane foam filling (dimensions 30 cm width x 50 cm length x 32 cm depth) for their transport and subsequent incubation. Since artificial incubation at site before the project did not include the collection of yolkless eggs for incubation, and since a real function related to optimizing the clutch environment has not been proved (Wallace et al. 2006 and 2007) we decided to keep the same practice excluding the yolkless eggs from the boxes.

We tried to mimic the natural clutch arrangement as much as possible placing yolked eggs in the boxes using the sand from the original nesting beach. Two to three centimetres of sand were placed at the bottom and sides of the box to prevent contact of eggs with box walls, as well as over the eggs at the top of the box.

To record hourly clutch temperatures, we placed TinyTag Plus-2 dataloggers (Gemini Data Loggers UK Ltd., Model TGP-4017, accurate to ± 0.3°C) in artificially incubated clutches in EB (n = 12) and WB (n = 33). An additional two loggers were placed as controls at nest depth in Mosquea and Bahía de las Águilas beaches in 2008 and 2009 but these were, unfortunately, lost to beach erosion.

When clutches were left *in situ*, we camouflaged the tracks of females and recorded GPS coordinates. We recorded distance from nest to the high tide line, and the beach zone where the clutch was located (open beach, at the vegetation border or within the dune vegetation). Once the clutches were camouflaged we estimated the date of hatching based on the average known incubation duration estimated of leatherback marine turtles (~ 60 days), and from 10 days before this date we checked the nest place in order to find hatchlings tracks.

Study of clutches

During the incubation, each box was checked daily for signs of hatched turtles. When artificially incubated clutches hatched, we carefully excavated them no earlier than 48 hours after the last sign of hatchling emergence. We excavated *in situ* clutches by hand one to three days after hatchlings emergence was detected in surveys. In both cases, data were collected on the number of hatched shells, live and dead hatchlings within the clutch. Clutch size was defined as the total number of eggs (hatched and unhatched). Hatching success was calculated by dividing the number of eggs hatched by the clutch size expressed as percentage (Miller 1999). We opened and examined the contents of all unhatched eggs. Eggs that had partial calcification of the shell or evidence of an early stage embryo were classified as early stage embryonic death (Bell et al. 2003). Eggs that contained dead hatchlings at a late stage of development were classified as late stage embryonic death.

For each clutch we calculated incubation duration (defined as the number of days between egg laying and the time of first hatchling emergence). Incubation was considered completed when hatchlings emerged on the top of the sand. A regression test was carried out to describe the relationship between incubation duration and mean temperature during the mid-third of incubation which is thought to contain the thermosensitive period (the time span outside of which temperature manipulations do not exert any influence on sexual phenotype, Mrosovsky and Pieau 1991). We estimated hatchling sex ratios using the conversion curve relating incubation duration to hatchling sex ratio derived from *in situ* incubated eggs originating from Suriname (Godfrey 1997). To apply this (Godfrey 1997) curve for artificially incubated clutches, we added 4.1 days to our incubation duration data, (Godfrey and Mrosovsky 1997). Examples of the use of this method and its validation are given in Zbinden et al. (2007) and Katselidis et al. (2012).

Statistical analysis

To understand the effects of incubation methods and nest metrics, we analyzed data using Generalized Linear Mixed Models (GLMMs) which allow both fixed and random factors as well as covariates to be fitted, and random factors control for the use of repeated measurements (Schall 1991). Our data were subject to temporal pseudoreplication caused by repeated measurements through time of the same females nesting over the season. We therefore included clutch and year as random effects in the models to account for the pseudoreplication among females and potential variation in the response variable across years. GLMMs were performed with the lme4 package (Bates et al. 2008) in R (version 2.14.0) and the graphical output was produced with the sciplot package (Morales 2012).

We examined the effect of incubation method (artificial incubation at eastern beaches (EB), artificial incubation at western beaches (WB) and *in situ*) on hatching success using GLMM analyses with a binomial error, the most appropriate error distribution for percentages data, with a logit link function. Random effects considered were year and clutch and the covariates included were: incubation method, incubation duration (days), clutch size (number of yolked eggs) and laying date.

We investigated how incubation method affected the frequency of late and early stage dead embryos within studied clutches. We used subsequent GLMMs with Poisson error, the most appropriate distribution for count data, and a log link function to examine the effect of incubation method on number of early stage dead embryos and late stage dead embryos occurring in clutches as in Pintus et al. (2009). Maximal models included clutch size (yolked eggs), incubation duration (days) and laying date as covariates as well as incubation method.

To analyze the effect of incubation method on incubation duration we performed GLMM with a Poisson error and a log-link function; with the maximal model also including incubation method, laying date and clutch size as covariates.

For each analysis we started from a maximal model with all fixed effects and interactions. Significance of fixed effects was assessed using likelihood-ratio tests (comparing against χ2 distribution) following deletion from the model until only those terms that explained significant deviance remained, starting with interactions (e.g., Weber et al. 2011).

Data exploration techniques were applied as described in Zuur et al. (2010) including multipanel scatterplots to look at the presence of collinearity (correlation between explanatory variables). All two-way interactions were tested, but results are only presented if found to explain significant variation. Homogeneity and heteroscedasticity were assessed based on a graphical inspection of the residuals (Zuur et al. 2009).

RESULTS

We studied a total of 109 clutches laid over the study period (2008-2009): 35 incubated artificially in eastern beaches (EB), 31 artificially incubated clutches in western beaches (WB) and 43 clutches incubated *in situ* at western beaches. We could not find 19 of the 62 clutches left *in situ*, likely due to erosion rainfall and floods. Table 5.1 shows descriptive statistics of number of yolked eggs, hatching success and incubation duration of clutches. We used data from 2008 and 2009 only to assess the effects of artificial incubation as no clutches were artificially incubated in WB in 2010.

Table 5.1. Mean (± SD) number of yolked eggs per clutch (YE), hatching success (HS %) and incubation duration (ID days) per clutch of leatherback turtles incubated under artificial (eastern and western beaches) and natural (*in situ*) conditions in Jaragua National Park during the four years of study. Range in parentheses, n = sample size.

	Eastern beaches				Western beaches				*In situ*			
Year	YE	HS	ID	FEMALE %	YE	HS	ID	FEMALE %	YE	HS	ID	FEMALE %
2008	65.1±18.4 (43-104) n = 12	53.1±16.4 (34.3-79.1) n = 12	72.6±6.1 (66-79) n = 5	2.5±3.8 (0-8) n = 5	76.3±15.1 (47-101) n = 22	42.9±23.9 (4.8-85.5) n = 22	62.7±2.1 (60-67) n = 20	41.7±23.5 (8-75) n = 20	71.1±14.8 (47-110) n = 22	85.3±19.9 (5.0-98.2) n = 22	64.9±1.8 (61-68) n = 14	53.6±28.5 (0-94) n = 14
2009	70.1±16.3 (24-107) n = 42	57.5±26.6 (0.0-93.6) n = 42	65.6±3.8 (59-74) n = 30	23.4±28.3 (0-85) n = 30	73.2±11.9 (54-92) n = 12	33.7±17.6 (6.0-55.8) n = 12	60.6±3.1 (55-64) n = 11	57.7±26.6 (21-99) n = 11	83.6±16.5 (56-120) n = 40	73.6±21.8 (12.3-97.9) n = 40	62.7±2.9 (57-70) n = 29	72.9±30.7 (0-100) n = 29
TOTAL	68.9±16.8 (24-107) n = 54	56.5±24.7 (0.0-93.6) n = 54	66.4±4.7 (59-79) n = 35	21.3±27.6 (0-85) n = 35	75.2±13.2 (47-101) n = 34	39.6±22.0 (4.8-85.5) n = 34	62.0±2.6 (55-67) n = 31	47.4±25.4 (8-99) n = 31	79.2±16.9 (47-120) n = 62	77.8±21.7 (5.0-98.2) n = 62	63.4±2.8 (57-70) n = 43	66.3±31.0 (0-100) n = 43

Effects of incubation method on hatching success

Incubation method significantly influenced hatching success (GLMM: $\chi^2 = 76.7$, $p < 0.001$; Figure 5.2a). *In situ* clutches had greater hatching success than those artificially incubated in WB and in EB (Table 5.1 and Figure 5.2a). We found clutch size (GLMM: $\chi^2 = 12.7$, $p < 0.05$), laying date (GLMM: $\chi^2 = 18$, $p < 0.01$) and incubation duration (GLMM: $\chi^2 = 20.1$, $p < 0.01$) exerted statistically significant effects on hatching success. We found incubation method to have an effect on the number of early stage embryo deaths (GLMM: $\chi^2 = 16.1$, $p < 0.05$; Figure 5.2b). Clutches artificially incubated in WB had more early stage dead embryos than *in situ* clutches and clutches artificially incubated in EB (Figure 5.2b). The number of late stage dead embryos in a clutch was also significantly affected by incubation method (GLMM: $\chi^2 = 67.4$, $p < 0.001$; Figure 5.2c). Artificially incubated clutches in EB and WB had more late stage dead embryos than in *in situ* clutches (Figure 5.2c).

Effect of incubation method on incubation duration

Across the two years, laying dates ranged from 09/03/2008 to 07/06/2008 and 25/03/2009 to 19/06/2009. Laying date was significantly negatively correlated with incubation duration in 2008 with longer incubation durations registered at the beginning of the season; however no correlation was found in 2009. Incubation method significantly influenced incubation duration (GLMM: $\chi^2 = 16.0$, $p < 0.01$). Clutches artificially incubated in EB had longer incubation duration than *in situ* clutches and clutches artificially incubated in WB (Figure 5.2d).

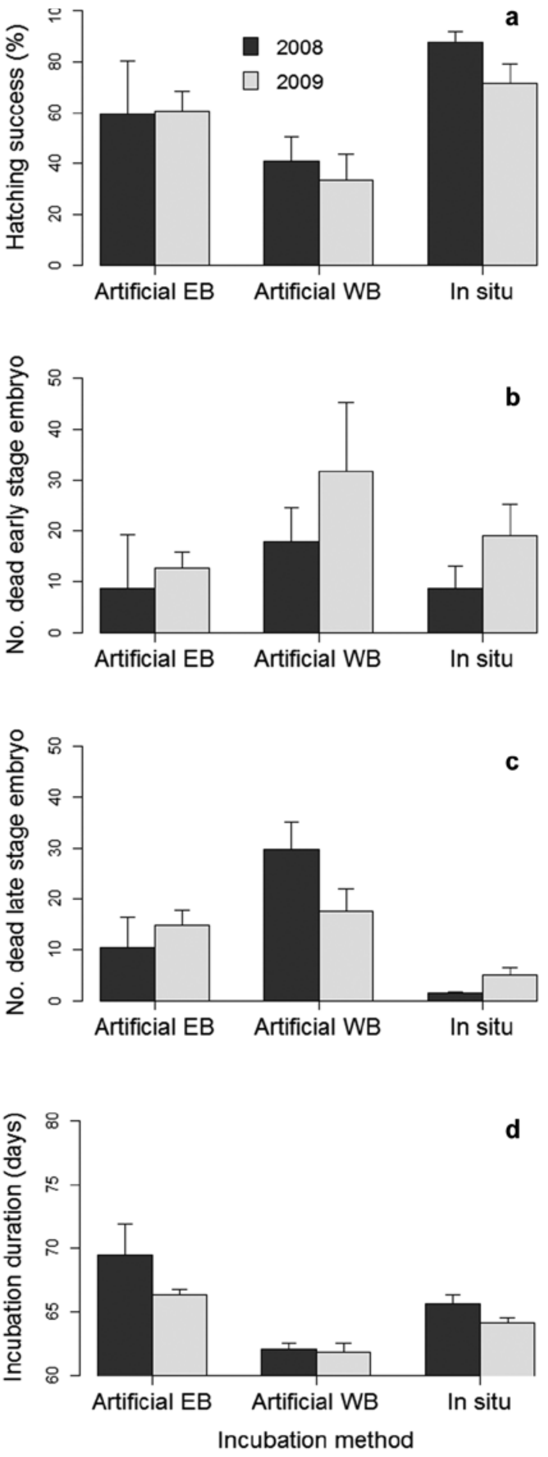

Figure 5.2. Effect of incubation method (artificially incubated eastern beaches (EB), western beaches (WB), and *in situ*) on a) hatching success.clutch^{-1}.year^{-1}, b) number of dead early stage embryos, c) number of dead late stage embryos, and d) incubation duration (days). Shown are fitted values of the model (Mean + SEM), which account for the effect of all significant predictor variables in the model and reflect the true relationship.

Mean incubation temperatures during the whole incubation duration were significantly lower in artificially clutches incubated at EB when compared to artificially incubated clutches at WB (Welch's t-test, $t_{22.9} = - 5.66$, $P < 0.001$; Figure 5.3a). Artificially incubated clutches at WB underwent large cycles of diel temperature variation and in some cases exceed the thermal tolerance range for embryo development under laboratory conditions (35ºC, Ackerman 1997) (Figure 5.3b). Incubation duration was correlated with mean temperature during the thermosensitive period ($F_{1, 42} = 15.9$, $P < 0.001$; Figure 5.4) thus it can be used to estimate sex ratio (Broderick et al. 2000).

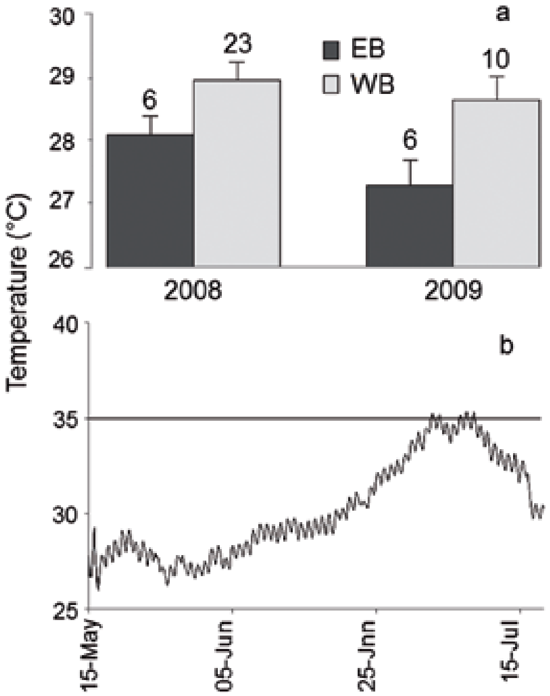

Figure 5.3. a) Temperatures for the whole incubation period (Mean + SEM) recorded in artificially incubated clutches at the two sites (EB: eastern beaches, and WB: western beaches) in 2008 and 2009, and b) hourly recorded incubation temperature profile of one clutch laid on 15 May 2008 and artificially incubated in WB that reached a maximum temperature of 35.3ºC and had 4.8% of hatching success. Solid line indicates 35ºC; the upper limit of tolerance range for marine turtle embryo development *in situ* (Ackerman, 1997). Number above bars indicates sample size.

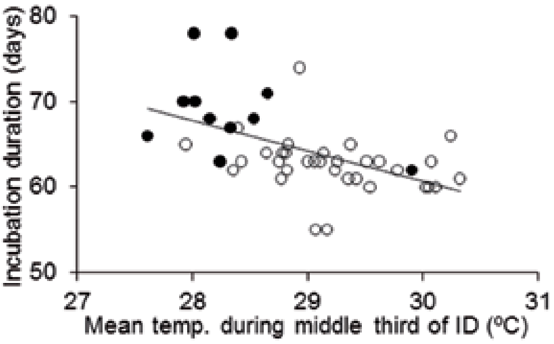

Figure 5.4. Relationship between incubation duration (ID) and mean temperature during the middle third of the incubation duration for 11 artificially incubated clutches in EB (solid circles) and for 33 artificially incubated clutches in WB (open circles). Equation: $y = - 3.06x + 152$, $r^2 = 0.28$.

DISSCUSION

Effects of artificial incubation on hatching success

The results of our analyses revealed that artificial incubation carried out to protect leatherback turtles clutches in Jaragua National Park caused changes both in hatching success and sex ratio, when compared to *in situ* clutches. Artificially incubated clutches, especially those incubated in boxes at western beaches (WB), exhibited lower hatching success than *in situ* incubated clutches. It is possible that hatching success in some artificially incubated clutches were affected by physical handling of eggs during relocation and transportation (Chan and Liew 1996), particularly in artificially incubated clutches of WB because of it was necessary to transport clutches further. However, we believe that the reduced hatching success in artificially incubated clutches was likely due to incubation conditions. Restricted exchange of heat, O2 and CO2 in boxes increases embryonic mortality, these differences in gas exchange might explain differences observed in incubation results (Ackerman 1980; Garret et al. 2010). Most eggs that failed to hatch in *in situ* clutches contained early stage embryos, mirroring trends seen in other leatherback populations (Eckert and Eckert 1990; Bell et al. 2003) whereas in artificially incubated clutches most eggs contained late stage embryos and many of them appeared to have died just prior to hatching, especially in WB. Metabolic heat generated by embryo activity (Wallace et al. 2004) could have caused this high late embryo mortality in WB because several boxes reached extremely incubation temperatures (above 35ºC), which can reduce hatching success (Santidrián Tomillo et al. 2009).

Although the function of yolkless eggs remains unclear (Eckert et al. 2012), recent studies have demonstrated that they could be 'production over-runs' of oviducts producing copious albumen for egg clutches and do not have a function related to facilitate gas exchange or temperature control within the clutch environment (Wallace et al. 2006 and 2007). However, Caut et al. (2006) showed that yolkless eggs had a positive effect on the clutch protecting yolk eggs from insect predation and Patiño-Martinez et al. (2010) propose that space released by dehydration of yolkless eggs may favour neonate emergence. Artificial incubation in boxes protects from predation and we kept hatchlings on the sand surface inside the box until they enter into frenzy before being released. For these reasons, and since other translocation programs have not found a significant difference in hatching success between clutches incubated with or without yolkless eggs (Dutton and McDonald 1995; Dal Pont Morisso and Krause 2004), we believe that not including yolkless eggs in artificially incubated clutches did not markedly affect hatching success.

Effects of artificial incubation on incubation duration

As well as impacting hatching success, the artificial incubation treatments also had an influence upon incubation temperature. The importance of incubation temperature is that it determines incubation duration and hatchling sex (Morreale et al. 1982). The longer incubation durations recorded in EB clutches could be due to environmental conditions, since the rangers' barrack where the clutches were incubated was located in a place with higher humidity and likely lower temperatures than in the beach. These longer incubation durations resulted in lower percentage of females produced in EB. However, since no clutches were incubated *in situ* in EB we could not find if there were significant differences on likely sex ratios produced by artificially incubated versus *in situ* clutches. Taking into account that sand color and quality can have an effect into thermal conditions on beaches (Naro-Maciel et al. 1999; Weber et al. 2011) it is possible that there are differences in natural sex ratio between beaches. The incubation site at WB is well away from the beach in an area of arid scrub and clutches reached high temperatures, resulting in similar incubation durations and sex ratios than *in situ* clutches.

It is important to note that eastern beaches are subject to ~100% illegal take of all clutches not relocated (Revuelta et al. 2012) and the shortage of human and economic resources prevents *in situ* clutch protection here. Therefore, clutch relocation is currently the only viable option for increasing recruitment, which has been recommended when illegal take of clutches approaches 100% (Mortimer 1999). We improved the previous program leaving the clutches incubating in the barracks of each beach and reducing transportation, replacing Styrofoam boxes by polyurethane ones and since 2011, we have been incubating 50% of EB clutches in a hatchery on the beach in order to mimic natural sex ratios as closely as possible.

For the artificially incubated clutches in WB, sex ratio proportions were similar to in *in situ* clutches but had lower hatching success. Western beaches are less frequented by walkers and rangers' facilities were installed there during the project, allowing us to reinforce protection and gradually reduce the number of clutches artificially incubated along the study period. In 2010, considering the low predation rate of *in situ* clutches we left all clutches incubating on the beach.

The present study fills the gap of leatherback turtle reproductive data in the Dominican Republic (Dow and Eckert 2011). In the eastern beaches of the Jaragua National Park the overall production of hatchlings incubated in boxes during the study period was much greater than would have been expected without protection efforts, but this method probably altered natural sex ratios. Hence, further research is needed to improve conservation measures at these beaches, including enforcement of beach protection and the development of an effective hatchery. Our results also indicated that artificial incubation at WB of JNP was an ineffective conservation strategy according to the low levels of hatching success recorded; thus current conservation is based in *in situ* clutch protection. However, WB are subject to unregulated tourism activity and development and are under the threat of the expansion of bauxite and limestone mining (Wielgus et al. 2010). Hence, this conservation strategy will be only possible with an institutional control over the influx of tourism and coastal development.

It is possible that, in the future, relocation and artificial incubation will be an option to avoid the extirpation of threatened marine turtle nesting populations due to habitat loss by coastal development or to the predicted rise of sea level. Nonetheless, our findings highlight the importance of monitoring and assessing mortality and sex ratio in conservation programs using these methods.

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REFERENCES

Ackerman RA (1980) Physiological and ecological aspects of gas exchange by sea turtle eggs. American Zoologist, 20, 575 - 583.

Ackerman RA (1997) The nest environment and the embryonic development of sea turtles. In: Lutz PL and Musick JA (eds). The biology of sea turtles. CRC Press Inc, Boca Raton FL pp 83 - 106.

Andrews HV (2000) Current marine turtle situation in the Andaman and Nicobar islands an urgent need for conservation action. Kachhapa, 3, 19 - 23.

Bates DM, Maechler M and Dai B (2008) Lme4: Linear mixed-effects models using S4 classes. http://lme4.r-forge.r-project.org/

Bell BA, Spotila JR, Paladino FV and Reina RD (2003) Low reproductive success of leatherback turtles, *Dermochelys coriacea*, is due to high embryonic mortality. Biological Conservation, 115, 131 - 138.

Broderick AC, Godley BJ, Reece S and Downie JR (2000) Incubation periods and sex ratios of green turtles: highly female biased hatchling production in the eastern Mediterranean. Marine Ecology Progress Series, 202, 273 - 281.

Broderick AC, Godley BJ and Hays GC (2001) Metabolic heating and the prediction of sex ratios for green turtles (*Chelonia mydas*). Physiological and Biochemical Zoology, 74, 161 - 170.

Carreras C, Godley BJ, León YM, Hawkes LA, Revuelta O, Raga JA and Tomás J (2013) Contextualising the last survivors: population structure of marine turtles in the Dominican Republic. PLoS ONE. doi:10.1371/journal.pone.0066037.

Caut S, Guirlet E, Jouquet P and Girondot M (2006) Influence of nest location and yolkless eggs on the hatching success of leatherback turtle clutches in French Guiana. Canadian Journal of Zoology, 84, 908 - 815.

Chan EH and Liew HC (1996) Decline of the leatherback population in Terengganu, Malaysia, 1956-1995. Chelonian Conservation and Biology, 2, 196 - 203.

Dal Pont Morisso E and Krause L (2004) How nest translocation-time, clutch size and presence of yolkless eggs affected hatching success in *Dermochelys coriacea* (Linnaeus, 1766) (testudines: dermochelyidae), at projeto Tamar-Ibama, Espírito Santo, Brazil. Cuadernos de Herpetología, 18, 59 - 64.

Dow WE and Eckert KL (2011) Sea turtle nesting habitat in the Wider Caribbean Region. Endangered Species Research, 15, 129 - 141.

Dow W, Eckert K, Palmer M and Kramer P (2007) An atlas of sea turtle nesting habitat for the Wider Caribbean Region. WIDECAST Technical Report No. 6. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, Beauford, USA.

Dutton PH, Whitmore CP and Mrosovsky N (1985) Masculinisation of leatherback turtle *Dermochelys coriacea* hatchlings from eggs incubated in Styrofoam boxes. Biological Conservation, 31, 249 - 264.

Dutton PH and McDonald D (1995) Hatch rates of leatherback (*Dermochelys coriacea*) clutches reburied with and without yolkless eggs. Proceedings of the 12th Annual Symposium on Sea Turtle Biology and Conservation.U.S. Dept. Commerce. NOAA Tech. Memo. NMFS-SEFSC-361, Jekyll Island, Georgia, EEUU, pp 177 - 178.

Dutton DL, Dutton PH, Chaloupka M and Boulon RH (2005) Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long term nest protection. Biological Conservation, 126, 186 - 194.

Eckert KL, Wallace BP, Frazier JG, Eckert SA and Pritchard PCH (2012) Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication BTP-R4015-2012, Washington, D.C.

Eckert KL and Eckert SA (1990) Embryo mortality and hatch success in in situ and translocated leatherback sea turtle *Dermochelys coriacea* eggs. Biological Conservation, 53, 37 - 46.

Foley AM, Peck SA, Harman GR and Richardson LW (2000) Loggerhead turtle (*Caretta caretta*) nesting habitat on low-relief mangrove islands in southwest Florida and consequences to hatchling sex ratios. Herpetologica, 56, 433 - 445.

Frazier J (2002) Marine turtles and international instruments: the agony and the ecstasy. Journal of International Wildlife Law and Policy, 5, 1 - 10.

Garret K, Wallace BP, Garner J and Paladino FV (2010) Variations in leatherback turtle nest environments: consequences for hatching success. Endangered Species Research, 11, 147 - 155.

Godfrey MH (1997) Sex ratios of sea turtle hatchlings: direct and indirect estimates. Phd Thesis. University of Toronto.

Godfrey MH and Mrosovsky N (1997) Estimating the time between hatching of sea turtles and their emergence from the nest. Chelonian Conservation and Biology, 2, 581 - 585.

Hamman M, Godfrey MH, Seminoff JA, Arthur K, Barata PCR, Bjorndal KA, Bolten AB, Broderick AC, Campbell LM, Carreras C, Casale P, Chaloupka M, Chan SKF, Coyne MS, Crowder LB, Diez CE, Dutton PH, Epperly SP, FitzSimmons NN, Formia A, Girondot M, Hays GC, Cheng IJ, Kaska Y, Lewison R, Mortimer JA, Nichols WJ, Reina RD, Shanker K, Spotila JR, Tomás J, Wallace BP, Work TM, Zbinden J and Godley BJ (2010) Global research priorities for sea turtles: informing management and conservation in the 21st century. Endangered Species Research, 11, 245 - 269.

IUCN (2012) The IUCN red list of threatened species (WWW document). URL: http://www.iucnredlist.org [accessed 19 November 2012].

Katselidis KA, Schofield G, Stamou GP, Dimopoulos P and Pantis JD (2012) Females first? Past, present and future variability in offspring sex ratio at a temperate sea turtle breeding area. Animal Conservation, 15, 508 - 18.

Liles MJ, Jandres MV, López WA, Mariona GI, Hasbún CR and Seminoff JA (2011) Hawksbill turtles *Eretmochelys imbricata* in El Salvador: nesting distribution and mortality at the largest remaining nesting aggregation in the eastern Pacific Ocean. Endangered Species Research, 14, 23 - 30.

McGowan A, Broderick AC, Frett G, Gore S, Hastings M, Pickering A, Wheatley D, White J, Witt MJ and Godley BJ (2008) Down but not out: marine turtles of the British Virgin Islands. Animal Conservation, 11, 92 - 103.

Miller JD (1999) Determining clutch size and hatch success. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ Species Survival Commission Marine Turtle Specialist Group Publication No. 4, Blanchard, USA, pp 124 - 129.

Morales M (2012) With code developed by the R Development Core Team and with general advice from the R-help listerv community, especially Duncan Murdoch. Sciplot: Scientific Graphing Functions for Factorial Designs. R package version 1.1-0.

Morreale SJ, Ruiz GJ, Spotila JR and Standora EA (1982) Temperature-dependent sex determination: current practices threaten conservation of sea turtles. Science, 216, 1245 - 1247.

Mortimer JA (1999) Reducing threats to eggs and hatchlings: hatcheries. In: Eckert, KL Bjorndal, KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ Species Survival Commission Marine Turtle Specialist Group Publication No. 4, Blanchard, USA, pp 175 - 178.

Mrosovsky N (1982) Sex ratio bias in hatchling sea turtles from artificially incubated eggs. Biological Conservation, 23, 309 - 314.

Mrosovsky N and Pieau C (1991). Transitional range of temperature, pivotal temperatures and thermosensitive stages for sex determination in reptiles. Amphibia-Reptilia, 12, 169 - 179.

Naro-Maciel E, Mrosovsky N and Marcovaldi MA (1999). Thermal profiles of sea turtle hatcheries and nesting areas at Praia do Forte, Brazil. Chelonian Conservation and Biology, 3, 407 - 413.

Ottenwalder JA (1981) Estudio preliminar sobre el estado, distribución, y biología reproductiva de las tortugas marinas en la República Dominicana. Tesis de Licenciatura, Universidad Autónoma de Santo Domingo, Dominican Republic.

Özdemir B and Türkozan O (2006) Hatching success of green turtle, *Chelonia mydas* clutches in relation biotic and abiotic factors in Northern Cyprus. Turkish Journal of Zoology, 30, 377 - 381.

Patiño-Martinez J, Marco A, Quiñones L and Godley BJ (2008) Globally significant nesting of the leatherback turtle (*Dermochelys coriacea*) on the Caribbean coast of Colombia and Panama. Biological Conservation, 141, 1982 - 1988.

Patiño-Martinez J, Marco A, Quiñones L and Calabuig C (2010) Los huevos falsos (SAGs) facilitan el comportamiento social de emergencia en las crías de la tortuga laúd *Dermochelys coriacea* (Testudines: Dermochelyidae). Revista de biología tropical, 58, 943 - 954.

Phillott AD and Parmenter CJ (2007) Deterioration of green (*Chelonia mydas*) eggs after known embryo mortality. Chelonian Conservation and Biology, 6, 262 - 266.

Pintus KJ, Godley BJ, McGowan A and Broderick AC (2009) Impact of clutch relocation on green turtle offspring. Journal of Wildlife Management, 73, 1151 - 1157.

Pullin AS and Knight TM (2003) Support for decision making in conservation practice: an evidence-based approach. Journal for Nature Conservation, 11, 83 - 90.

Pullin AS and Knight TM (2009) Doing more good than harm: building an evidence-base for conservation and environmental management. Biological Conservation, 142, 931 - 934.

Revuelta O, León YM, Feliz P, Godley BJ, Raga JA and Tomás J (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. Oryx, 46, 348 - 358.

Santidrián Tomillo P, Saba V, Piedra R, Paladino FV and Spotila JR (2008) Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. Conservation Biology, 22, 1216 - 1224.

Santidrián Tomillo P, Suss JS, Wallace BP, Magrini KD, Blanco G, Paladino FV and Spotila JR (2009) Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology, 156, 2021 - 2031.

Sarti Martínez L, Barragán AR, García Muñoz D, García N, Huerta P and Vargas F (2007) Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology, 6, 70 - 78.

Schall R (1991) Estimation in generalized linear models with random effects. Biometrika, 78, 719 - 727.

Schroeder B and Murphy S (1999) Population surveys (ground and aerial) on nesting beaches. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ Species Survival Commission Marine Turtle Specialist Group Publication No. 4, Blanchard, USA, pp 45 - 53.

Spotila JR, Reina RD, Steyermark AC and Plotkin PT (2000) Pacific leatherback turtles face extinction. Nature, 405, 529 - 530.

Sutherland WJ, Adams WM, Aronson RB, Aveling R, Blackburn TM, Broad S, Ceballos G, Coté IM, Cowling RM, Da Fonseca GAB, Dinerstein E, Ferraro PJ, Fleishman E, Gascon C, Hunter M JR, Hutton J, Kareiva P, Kuria A, Macdonald DW, Mackinnon K, Madgwick FJ, Mascia MB, McNeely J, Milner-Gulland EJ, Moon S, Morley CG, Nelson S, Osborn D, Pai M, Parsons ECM, Peck LS, Possingham H, Prior SV, Pullin AS, Rands MRW, Ranganathan J, Redford KH, Rodriguez JP, Seymour F, Sobel J, Sodhi NS, Stott A, Vance-borland K and Watkinson AR (2009) One hundred questions of importance to the conservation of global biological diversity. Conservation Biology, 23, 557 - 567.

Tuttle J and Rostal D (2010) Effects of nest relocation on nest temperature and embryonic development of loggerhead sea turtles (*Caretta caretta*). Chelonian Conservation and Biology, 9, 1 - 7.

Wallace BP, Sotherland PR, Spotila JR, Reina RD, Franks BF and Paladino FV (2004) Biotic and abiotic factors affect the nest environment of embryonic leatherback turtles, *Dermochelys coriacea.* Physiological and Biochemical Zoology, 77, 423 - 432.

Wallace BP, Sotherland PR, Santidrian Tomillo P, Bouchard SS, Reina RD, Spotila JR and Paladino FV (2006) Egg components, egg size, and hatchling size in leatherback turtles. Comparative Biochemistry and Physiology, 145, 524 - 532.

Wallace BP, Sotherland PR, Tomillo PS, Reina RD, Spotila JR and Paladino FV (2007) Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia, 152, 37 - 47.

Weber S, Broderick AC, Groothuis TGG, Ellick J, Godley BJ and Blount JD (2011) Fine-scale thermal adaptation in a green turtle nesting population. Proceedings of the Royal Society B, 279, 1077 - 1084.

Wielgus J, Cooper E, Torres R and Burke L (2010) Coastal Capital: Dominican Republic. Case studies on the economic value of coastal ecosystems in the Dominican Republic. Working paper. World Resources Institute, Washington, DC, USA.

Witt MJ, Bonguno EA, Broderick AC, Coyne MS, Formia A, Gibudi A, Mounguengui GA, Moussounda C, NSafou M, Nougessono S, Parnell RJ, Sounguet G, Verhage S and Godley BJ (2011) Tracking leatherback turtles from the world's largest rookery: assessing threats across the South Atlantic. Proceedings of the Royal Society B, 278, 2338 - 2347.

Yntema CL and Mrosovsky N (1982) Critical periods and pivotal temperatures for sexual differentiation in loggerhead turtles. Canadian Journal of Zoology, 60, 1012 - 1016.

Zbinden JA, Davy C, Margaritoulis D and Arlettaz R (2007) Large spatial variation and female bias in the estimated sex ratio of loggerhead sea turtle hatchlings of a Mediterranean rookery. Endangered Species Research, 3, 305 - 312.

Zuur AF, Leno EN and Meesters EHWG (2009) A Beginner's Guide to R. Springer, New York.

Zuur AF, Leno EN and Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods in Ecology & Evolution, 1, 3 - 14.

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CHAPTER VI:

RUNNING AGAINST TIME: CONSERVATION OF THE REMAINING HAWKSBILL TURTLE (*ERETMOCHELYS IMBRICATA*) NESTING POPULATION IN THE DOMINICAN REPUBLIC

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CHAPTER VI:

RUNNING AGAINST TIME: CONSERVATION OF THE REMAINING HAWKSBILL TURTLE (*ERETMOCHELYS IMBRICATA*) NESTING POPULATION IN THE DOMINICAN REPUBLIC

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ABSTRACT

Saona Island hosts the last hawksbill turtle (*Eretmochelys imbricata*) nesting population in the Dominican Republic, which has experienced a severe decline in the last decades, mostly due to illegal egg take. Here we present the results of an artificial incubation programme started in 2007 to protect the clutches from human predation. A preliminary survey in 2006 showed that about 60% of clutches laid were taken by humans. Over the study period (2007-2010) we recorded 400 clutches, of which 38.2% were predated by humans, 40.7% were artificially incubated and 21% were incubated *in situ*. Overall, the artificial incubation programme allowed the release of 12,340 hatchlings. No differences were found in hatching and emergence success between clutches incubated *in situ* and clutches artificially incubated. However, incubation temperatures and incubation durations recorded suggest a male-biased hatchling sex-ratio in artificially incubated clutches. Although artificial incubation may mitigate the effect of egg take, our results indicate that other measures, such as clutch relocation to protected sections of the beach should be taken. Beach patrolling and education are currently implemented so that artificial incubation will be eventually phased out in favour of *in situ* incubation.

INTRODUCTION

The hawksbill marine turtle, *Eretmochelys imbricata*, is listed under the IUCN Global Red List as critically endangered (IUCN 2011). In the Caribbean, this species has been exploited historically for its meat, eggs and shell (Meylan 1999; Carrillo et al. 1999; Fleming 2001) leading to a noticeable reduction in nesting populations in the region (Meylan 2002; McClenachan et al. 2006). However, the regional management unit of Caribbean hawksbill is currently characterized as low risk but as high threat (Wallace et al. 2010), and some populations are thought to be increasing (Puerto Rico: Meylan 1999; Antigua: Richardson et al. 2006; Barbados: Beggs et al. 2007; Guadeloupe: Kamel and Delcroix 2009). However, rookeries without protection in this basin are seriously threatened, particularly due to egg take (Lagueux and Campbell 2005).

Increasing hatchling production is a necessary component of any strategy to recover depleted marine turtle populations (Dutton et al. 2005; Sarti Martínez et al. 2007). When clutches are at risk, relocation to different sections of the beach or to protected hatcheries have been common management tools in many marine turtle conservation programmes (Pritchard 1995; Kornaraki et al. 2006; Marcovaldi et al. 2007). However, this is not a perfect solution, as it has been reported that relocated clutches may experience lower hatching success and altered sex-ratios (Godfrey and Mrosovsky 1999; Mortimer 1999; Pintus et al. 2009). Nevertheless, the intense pressures from human egg take in undeveloped countries leaves them with no other option than relocating clutches and using hatcheries (García et al. 2003; Liles et al. 2011) or even, in extreme situations, to incubate clutches in boxes (Dutton et al. 1985). Therefore, clutch conservation strategies should rely on the evaluation of local characteristics, and should be adapted to the specific scenarios where they are intended to be applied (Eckert 1999; Kornaraki et al. 2006).

In the Dominican Republic (DR, North Caribbean), current estimates suggest a severe decline of the hawksbill nesting population, with sporadic nesting in few places around the country and only one remaining stable stock on the south coast of Saona Island, south-east DR (with a mean of 23 females nesting per year: Revuelta et al. 2012). According to the estimation of 5000 females nesting annually in the Caribbean (Meylan 1999), Saona Island hawksbill stock would represent close to 0.5% of the total number of nesting females in the region. The greatest threat to this nesting rookery comes from illegal egg take and deliberate capture of adults for meat (Revuelta et al. 2012). Preliminary surveys in 2006 revealed that up to 60% of 41 nests recorded on the island were taken by humans, with up to 100% (n = 13) taken near the principal human settlement, Mano Juan (Revuelta et al. unpublished data). This occurs even though it is illegal to harvest turtles or their products in the DR since 1966, and that Saona is part of a protected area since 1975 (Del Este National Park). This situation could potentially impact rookeries on a regional scale since Dominican hawksbills seem to disperse to several feeding grounds throughout the Caribbean after nesting; including distant areas in waters of Nicaragua and Honduras (Hawkes et al. 2012).

It is well known that artificial incubation in boxes may introduce masculinizing biases in resulting hatchlings (Morreale et al. 1982; Dutton et al. 1985; Whitmore and Dutton 1985). This is because sex in marine turtles is determined by the temperature at which eggs are incubated during the middle third of incubation, which is referred to as the thermosensitive period (Yntema and Mrosovsky 1982). When boxes are used, the temperature during this period is often lower than pivotal temperature (that which produces 1:1 sex ratio: Mrosovsky and Pieau 1991), thus resulting in a greater proportion of male hatchlings (Mrosovsky 1994). However, given the high predation levels observed, in 2007, we initiated a programme of artificial incubation in boxes as an urgent measure to protect the clutches and increase hatchling recruitment while implementing other conservation measures, such as beach patrolling and education.

Here we present the results of a four year monitoring and conservation programme of the hawksbill nesting population of Saona Island, including the artificial incubation programme. Our main research questions were twofold: (1) are there differences in hatching and emergence success between artificially incubated clutches and *in situ* incubated clutches?; and (2) what is the sex-ratio produced in the artificially incubated clutches?

MATERIALS AND METHODS

Study area

The present study was conducted from 2007 to 2010 in Saona Island (18º07'N 68º44'W; Figure 6.1), which is included in the Del Este National Park (south-east DR). With an area of 110 km^2, Saona is the largest island adjacent to the DR. There is one permanent human settlement in Saona, Mano Juan village, with a population of 300 inhabitants (Figure 6.1). Hawksbill nesting occurs across all of the 26.6 km of the south coast sandy beaches of Saona Island. However, nesting activity is concentrated in 12 narrow fine-grained coralline, white sand beaches (altogether comprising a total of 15 km; Table 6.1) interspersed with rocky areas. The neritic sea adjacent to the nesting beach is composed of coral reefs and seagrass beds. For the most part, beach vegetation is dominated by coconut (*Cocos nucifera*) plantations, sea purslane (*Sesuvium portulacastrum*), sea rosemary (*Suriana maritima*), sea grape (*Coccoloba uvifera*), goat's foot creeper (*Ipomoea pes-caprae*) and native grasses.

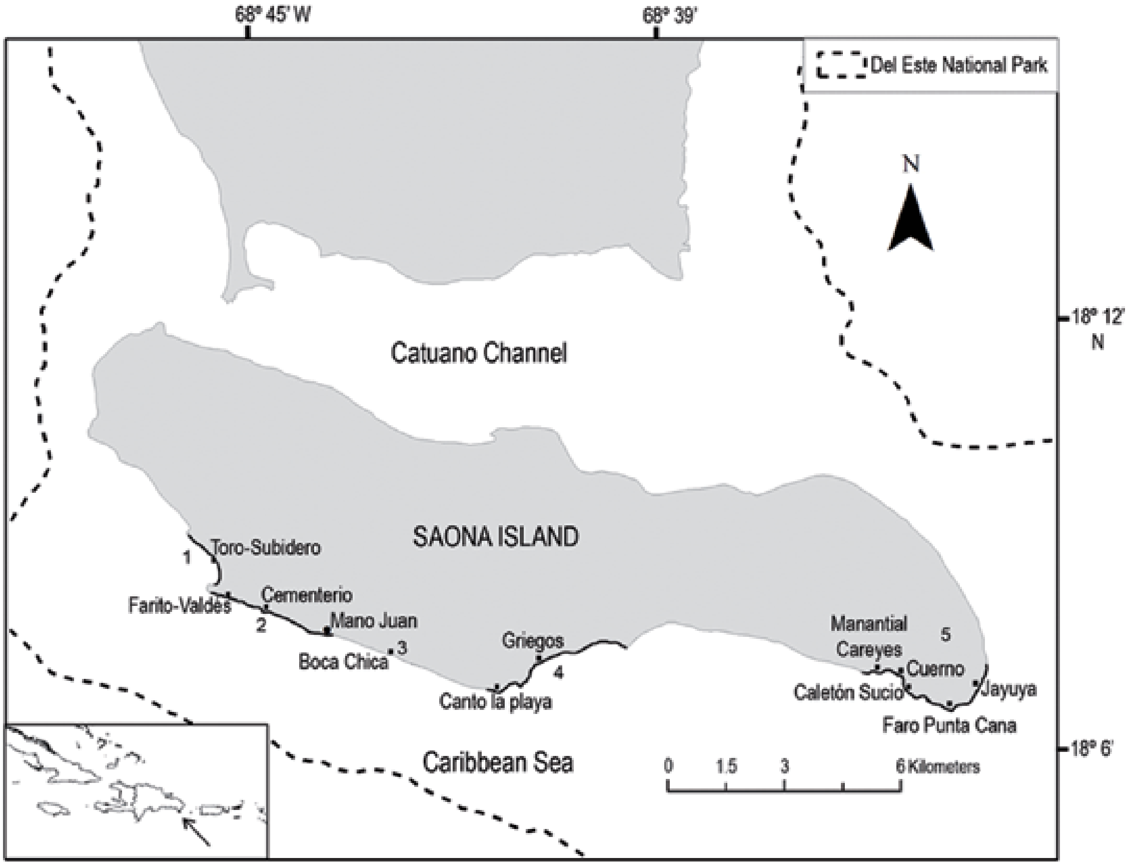

Figure 6.1. Hawksbill turtle nesting areas (n = 5) and beaches surveyed (n = 12) in Saona Island in Del Este National Park, Dominican Republic. Inserted in the map is shown the location of the study area within the Caribbean Sea.

Beaches were grouped into 5 sampling areas: (1) Del Toro, with 1530 m length and mean beach width (range) of 7.9 (4-18) m; (2) Mano Juan, 2990 × 8 (2-20) m; (3) Boca Chica, with 12.9 × 4.4 m; (4) Canto de la Playa area, 4263 × 18 (8-30) m; and (5) Faro Punta Cana, 6084 × 10 (3-20) m (Figure 6.1). We patrolled the beaches at least once per week throughout the year, but in 2008 this was increased to 3-4 surveys per week during the peak nesting period (June to November). Also, in 2009, the farthest area (Faro Punta Cana; see Figure 6.1) was visited only monthly due to logistic limitations.

We patrolled the beaches, by foot to detect all recent tracks of nesting females. All emergences of turtle females to nest were recorded. Clutches were confirmed by the presence of eggs in the nest chamber, or by signs of egg predation at the nesting site (see below). For each nest we recorded its GPS location, and position on the beach (open sand, vegetation border and within vegetation). We measured minimum distances from the nest to the high tide line. We classified nests as intact (eggs in nest chamber), or predated (no eggs in nest chamber, presence of probing sticks, eggshells and human footprints littering the nesting site). Owing to high levels of human egg take, the majority of clutches detected (n = 163) were removed for artificial incubation. In order to investigate the potential effects of artificial incubation on hatching and emergence success we studied 49 clutches incubated *in* *situ*, using them as controls. To reduce predation, all nests left *in situ* were camouflaged by erasing turtle tracks and other signs of nesting activity.

Clutches moved for artificial incubation were carefully excavated by hand, and the eggs were relocated into polyethylene exterior boxes with polyurethane foam filling. Two to three centimetres of beach sand was put at the bottom and sides of the box to prevent contact of eggs with walls. We placed a TinyTag temperature logger (Gemini Data Loggers UK Ltd., Model TGP-4017; accuracy +0.38C) inside a subsample of 22 artificially incubated clutches. Loggers were programmed to record temperatures at sampling intervals of 1 hour throughout the incubation duration. Extra care was taken when moving clutches, limiting shake and vibrations to avoid dislodging the embryo (Chan 1989; Almeida and Mendes 2007). Each box was labelled with a code indicating beach name, the laying date and number of eggs incubated. Boxes were placed in a facility located about 50 m from the beach of Mano Juan village. This facility (4 × 3 m (length × width)) had a sand-beach floor, walls made of wire mesh and corrugated metal panels, and roof made from palm leaves to protect boxes against rain and flooding. The boxes were checked daily throughout the incubation duration and lids opened for two to three hours a day to allow air to circulate in the otherwise air-tight box. Hatchlings were released at sunset one day after they emerged when they enter into frenzy.

To detect effects on hatchling fitness that might be due to artificial incubation, we compared the size and weight of hatchlings produced in boxes with those of hatchlings from other nesting rookeries in the Caribbean (e.g. Mona island, Puerto Rico: van Dam and Diez 1998; Cuba: Moncada et al. 1999; British Virgin Islands: Hillis and Phillips 1996). Random samples of 20 hatchlings each from 24 artificially incubated clutches in 2008 and 8 artificially incubated clutches in 2009 were measured (straight carapace length (SCL)) to the nearest 0.1 cm with a calliper, and weighed to the nearest 0.1 g with an electronic scale.

Artificially incubated and *in situ* clutches were studied, and clutch size, hatching and emergence success were considered according to definitions in Miller (1999). In this study, it could not be known in advance whether the clutches we collected for artificial incubation were a random subset of all the clutches on the beach. A way to shed light on this question could be to check whether clutch size differed between artificially incubated and *in situ* clutches (note that, in the latter, clutch size could only be determined after emergence of hatchlings). A two-way analysis of variance (ANOVA) revealed that the mean number of eggs per clutch did not differ with incubation type ($F_{(1,157)} = 0.85$, $P = 0.357$) or among years ($F_{(3,157)} = 2.46$, $P = 0.06$).

We tested whether incubation type affected hatching success or emergence success. Since data were gathered in 4 years, we used a full factorial two-way ANOVA, with 'incubation type' (*in situ*, artificial) and 'year' as fixed and random factors, respectively. Before statistical testing, data about hatching and emergence success were arcsin-transformed to achieve normality and homoscedasticity (Sokal and Rohlf 1995).

For each clutch, incubation duration was defined as the number of days between egg-laying and the first emergence recorded. We used temperature data to provide an estimation of sex-ratio produced in the artificially incubated clutches. Since pivotal temperature has not been determined for hawksbill turtle in Saona, we used the published curve relating incubation temperature and hatchling sex-ratio derived from laboratory incubated eggs from the closest hawksbill nesting area at Mona Island, Puerto Rico (Mrosovsky et al. 2009). We read off the means of temperature recorded in the thermosensitive period of the 22 clutches on the Mona laboratory curve to estimate female proportions. Examples of the use of this method and its validation are given in Wibbels (2003) and Öz et al. (2004).

RESULTS

Table 6.1. Number of nests and distribution of hawksbill turtle nesting density per beach in Saona Island, during 2007-2010. BL, beach length (km); MANR, mean annual nests rate (nests.years^{-1}); ND, nesting density (nests.Years^{-1}.kms^{-1}).

Area	Beach	BL (km)	N	MANR	ND
Toro	Toro-Subidero	1.5	39	9.0	6.0
Mano Juan	Farito-Valdés	0.7	65	16.2	23.1
	Cementerio	0.29	52	13.0	44.8
	Mano Juan	2.0	18	4.5	2.2
Boca Chica	Boca Chica	0.01	23	5.7	438.4
Canto Playa	Canto Playa	1.2	66	16.5	13.7
	Griegos	3.1	11	2.7	0.8
F.Punta Cana	M. Careyes	0.8	6	1.5	1.8
	Cuerno	1.5	22	5.5	3.7
	Caletón Sucio	0.5	20	5.0	10.0
	F.Punta Cana	2.1	68	17.0	8.1
	Jayuya	1.2	10	2.5	2.1

Nesting density and nest site selection

Hawksbill turtles laid a mean of 100 nests.year^{-1} (± standard deviation (SD) 8.4: range 93 to 111) on Saona Island. The highest mean annual nesting rate occurred at Mano Juan area (33.7 nests.year^{-1}) followed by Faro Punta Cana area (31.5 nests.year^{-1}), Canto de la Playa area (19.2 nests.year^{-1}) and Del Toro area (9.0 nest.year^{-1}). The mean nesting linear density over the 15 km of available nesting habitat was 6.6 nests.year^{-1}.km^{-1}. Apart from Boca Chica, an isolated small beach surrounded of rocky areas, with an average of 5.7 nests.years^{-1}, the highest nesting density reported was on Mano Juan area (11.3 nests.year^{-1}.km^{-1}), followed by Del Toro area (6 nests.year^{-1}.km^{-1}), Faro Punta Cana (5.2 nests.year^{-1}.km^{-1}) and Canto de la Playa (4.5 nests.year^{-1}.km^{-1}) (see Table 6.1). We did not find significant correlation between beach width and number of clutches laid ($r_s = 0.451$, $P = 0.141$, $n = 12$).

Saona hawksbills laid their eggs mainly in zones with at least some vegetation (within vegetation: 26.5%, n = 75; vegetation border: 56.0%, n = 158; open sand: 17.4%, n = 49). Overall, the mean distance from the nest to the high tide line was 8.9 m (SD = 5.3; range: 0.5-35 m). The beach with highest mean distance (16.7 m; SD = 7.0; range: 8-30) was Canto de la Playa beach, in Canto de la Playa area; and the beach with lowest mean distance (5.7 m; SD = 3.2; range: 2-13) was Cementerio beach, located in the Mano Juan area (Figure 6.1).

Predation levels and fate of clutches

In the four-year period we recorded a total of 400 hawksbill clutches in the 5 sampling areas. Considering the extensive experience detecting hawksbill tracks by the team and the low levels of nesting activity, we believe that weekly surveys allowed us to record close to 100% of clutches laid.

The majority of recorded clutches were artificially incubated (n = 163), while 146 clutches had already been predated by humans when found and 91 were left *in situ* (Figure 6.2). Humans took an additional 7 of these 91 clutches left *in situ*, and other 15 clutches of them were affected by tropical storms. We could not find (likely due to erosion and wash-out) another 20 of the 91 clutches left *in situ*. These events left a total of 49 clutches incubated *in situ* for the study.

Hence, in total we include in the study 365 clutches of three different fates: (i) predated by humans (n = 153); (ii) incubated artificially (n = 163); and (iii) incubated *in situ* (n = 49). However, for the estimations of hatching and emergence success we only considered clutches that were studied by the authors (n = 119) and (n = 46) artificially incubated clutches and *in situ* clutches, respectively. Figure 6.3 shows the annual variation on the percentages of these categories. All the clutches that were not artificially incubated or camouflaged *in situ* were found predated by humans in all beaches of the five areas of study. When humans predated a clutch, 100% of eggs were always taken. No evidence of natural predation (i.e. by crabs, ants, or feral pigs) on eggs was observed during the study period.

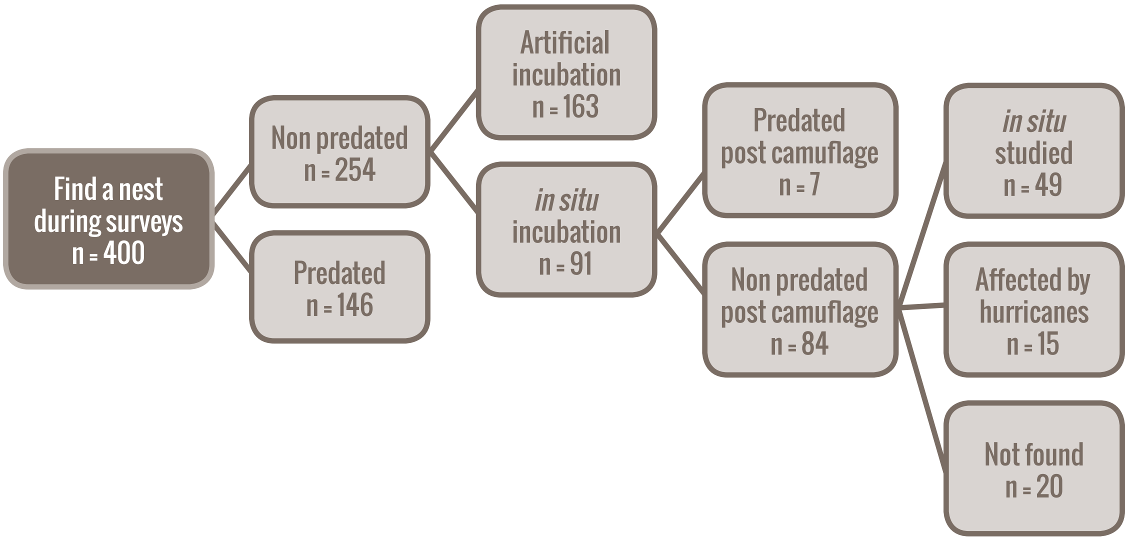

Figure 6.2. Clutch fate: number of clutches laid and their fate in Saona Island during the period 2007-2010. n, total number of clutches; predated, number of clutches predated by humans; artificial incubation, number of clutches removed for their incubation in boxes; *in situ* incubation, number of clutches camouflaged and left incubating on the beach; predated post-camouflage, number of clutches predated by humans after we camouflaged it; *in situ* studied, clutches found and studied after being camouflaged; affected by hurricanes, number of clutches incubating on beach when a hurricane made landfall in the island; not found, number of camouflaged clutches we could not find again due to beach dynamics.

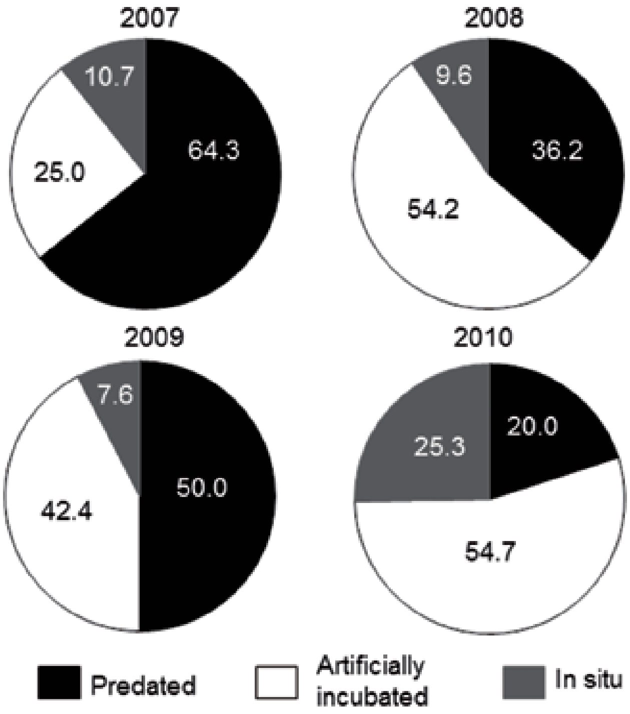

Figure 6.3. Percentages of clutches that were incubated artificially, clutches that were poached and clutches that hatched successfully *in situ* from the total of studied clutches every year (n = 84; n = 94; n = 92 and n = 95) respectively.

Hatching success and emergence success

Hatching and emergence success of 46 *in situ* incubated clutches and 119 artificially incubated clutches are shown in Table 6.2. We did not detect significant effects of artificial incubation in hatching success between years (two-way ANOVA: $F_{(3,157)}$ =1.45, P = 0.383), incubation type ($F_{(1,157)}$ = 0.7, P = 0.45); or their interaction ($F_{(3,157)}$ = 0.52, P = 0.669). Likewise, there was no significant difference in emergence success between years (two-way ANOVA: $F_{(3,157)}$ = 1.1, P = 0.469), incubation type ($F_{(1,157)}$ = 1.28, P = 0.327); or their interaction ($F_{(3,157)}$ = 0.68, P = 0.563) compared to *in situ* incubation.

In total, 12,340 hatchlings were produced under artificial conditions and released to the sea (1731 in 2007, 4522 in 2008, 2664 in 2009 and 3423 in 2010). Mean carapace length and weight of hatchlings was 3.8 ± 0.1 cm (range: 3.2-4.2) and 14.8 ± 1.2 g (range: 10.5-18.7) in 2008 (n = 480) and 3.8 ± 0.2 cm (range: 3.0-4.0) and 15.1 ± 1.1 g (range: 12.6-18.2) in 2009 (n = 160).

Table 6.2. Number of yolked eggs (YE), hatching success (HS), emergence success (ES) and incubation duration (ID) per clutch of hawksbill turtles reared under natural (*in situ*) and artificial conditions in Saona Island during the four years of study. N, number of clutches.

YEAR	N	YE	HS			ES			N	ID		
In situ			Range	Mean	SD	Range	Mean	SD		Range	Mean	SD
2007	9	1092	37.0-92.6	72.3	16.6	35.2-92.6	71.1	17.6	4	57- 64	60.6	3.4
2008	9	1392	5.1 - 96.5	73.3	30.9	5.1 - 96.5	72.1	30.3	4	68- 90	80.7	9.3
2009	7	929	52.8-91.4	78.3	12.8	34.9-91.4	72.0	21.0	0	–	–	–
2010	21	3034	7.1 - 98.3	71.8	25.5	7.1 - 98.3	67.0	25.7	16	50- 79	60.9	8.0
TOTAL	**46**	**6447**	**5.1 - 98.3**	**73.2**	**23.1**	**5.1 - 98.3**	**69.5**	**24.0**	**24**	**50- 90**	**64.2**	**10.6**
Artificial												
2007	20	2513	28.9-99.2	72.3	20.3	28.9-99.2	68.9	20.2	14	52- 66	59.8	3.7
2008	41	5667	44.5-98.5	81.8	13.9	32.1-98.5	79.8	14.7	25	53- 85	62.6	9.5
2009	25	3501	40.4-97.6	77.5	13.3	39.4-97.6	76.1	13.9	21	56- 71	59.4	4.1
2010	33	4588	27.6-100	76.7	20.6	27.6 -100	74.6	20.4	24	53- 85	62.3	8.4
TOTAL	**119**	**16269**	**27.6- 100**	**77.9**	**17.1**	**27.6- 100**	**75.5**	**17.6**	**84**	**52- 85**	**61.3**	**7.3**

Incubation temperature and incubation duration

Since beaches were not daily patrolled, recording the exact laying date was not possible in all cases. However, we were able to determine incubation duration for 84 out of the 119 artificially incubated clutches and 24 out of 46 *in situ* incubated clutches studied (Table 6.2). Temperatures in artificially incubated clutches ranged from 22.2 to 36.4°C, with mean values ranging from 26.8 to 30.2°C (n = 22). During the thermosensitive period temperatures recorded ranged from 22.2 to 33.8°C, with mean values ranging from 25.7 to 29.8°C. Incubation duration was strongly correlated with mean temperature during the thermosensitive period (F = 70.7, P, 0.001, df= 1; Figure 6.4).

In 21 of the 22 artificially incubated clutches mean temperature during the thermosensitive period was lower than the pivotal temperature derived from laboratory studies (29.6°C: Mrosovsky et al. 2009) (Figure 6.5), suggesting a male bias in artificially incubated clutches. The highest estimated female percentages (42% and 73%) were recorded in two clutches laid during the first half of August, and the estimated percentage of females was 0% in all artificially incubated clutches laid after the second half of August (Figure 6.5). Accordingly, we found a great intraseasonal variation in incubation duration in artificially incubated clutches every year, with longer incubation duration during colder months (October-December).

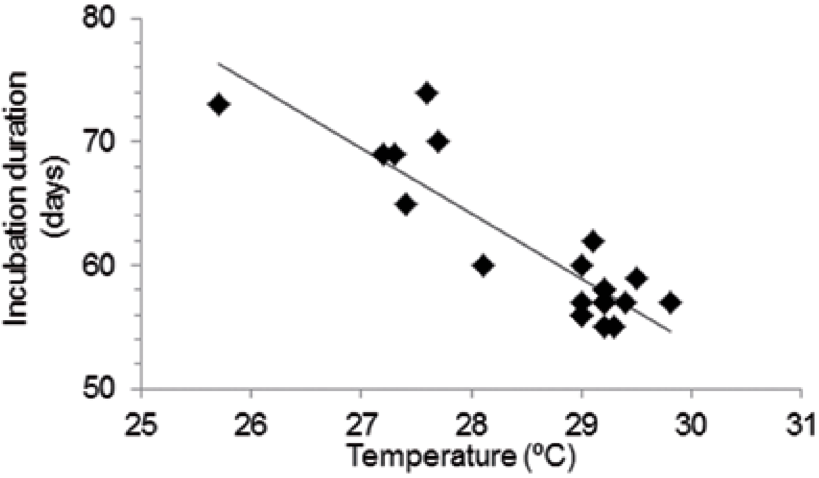

Figure 6.4. Relationship between incubation duration and mean temperature during the thermosensitive period in the 22 artificially incubated clutches controlled during the study (Equation: ID = 211-5.26 mean temperature during the thermosensitive period, r-square = 0.78).

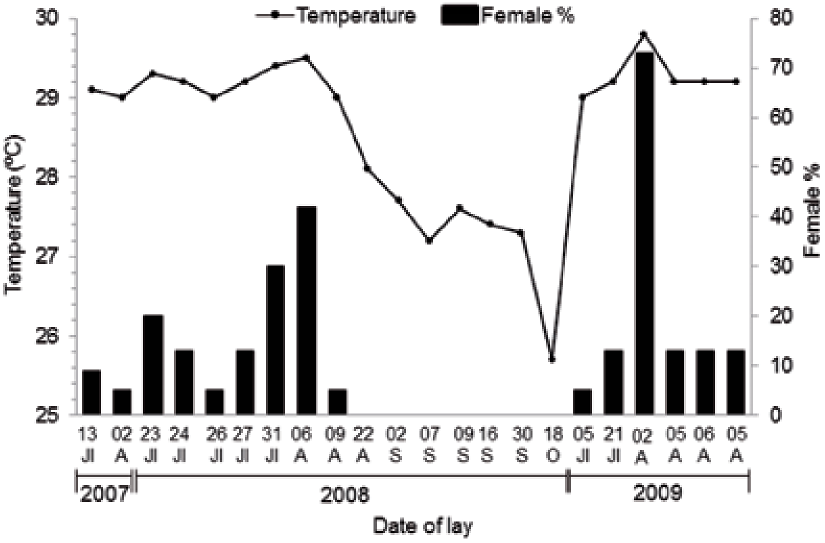

Figure 6.5. Laying date, mean incubation temperature during thermosensitive period (dots) and estimated female proportion (bars) of artificially incubated clutches (n = 22) in which temperature was monitored during the study period.

DISCUSSION

The present study provides, for the first time, detailed information about the reproductive biology of the remaining, threatened hawksbill nesting population of Saona Island, as well as the results of the conservation programme that we carried out to mitigate egg take.

Nesting density and nest site selection

Hawksbill females on Saona mainly nest in vegetated areas, this has also been noted in other studies of the species in other Caribbean rookeries (Pérez-Castañeda et al. 2007; Kamel and Delcroix 2009). Other authors have proposed that this could be because areas with vegetation might be less compact making hatchling emergence easier; also, the shade provided by vegetation might help keep temperature constant (Horrocks and Scott 1991; Kamel and Mrosovsky 2006). Moreover, the two beaches with the highest nest density in Saona (Boca Chica and Cementerio) have certain characteristics which make them suitable for nesting: an accessible vegetation area, a reef in front of the beach which protects them against surf and makes beach access easier, and fine grain sand that allows turtles digging their nests easily.

Predation levels and fate of clutches

Our results show the need to continue patrol and conservation work in Saona, since most *in situ* clutches not promptly camouflaged or relocated for artificial incubation, were predated by humans. In fact, in 2009, as a result of the decrease in surveys at the Faro Punta Cana area, 28 out of the 30 clutches laid were taken. In other words, the current level of egg take is unsustainable for the long term preservation of this nesting population.

Nesting beaches in Saona are scattered along the coastline and some of them can only be reached by boat, considerably raising monitoring costs. Also, the funding available did not allow the employment of personnel to make daily patrols over all the nesting beaches or nest surveillance. This left us with two options, i.e. relocating or camouflaging all clutches that were non-predated at the time of finding. Camouflage was considered to be an insufficient conservation measure since substantial amounts of camouflaged clutches were eventually predated by humans. The same situation has been described in other studies with similar conditions (Sato and Madriasau 1991; Lagueux et al. 2003). On the other hand, tropical storms that hit the island affected camouflaged clutches (4 affected by Noel in 2007, 7 by Hanna in 2008 and 4 by Earl and Fiona in 2010) making excavation and study impossible. It has been reported that excessive rainfall and floods can greatly reduce egg viability, thus reducing clutch survivorship (Martin 1996; Kamel and Delcroix 2009). Obviously, these threats are natural and unavoidable, but they add significant losses to the already depleted population. For all these reasons, and due to shortage of human and economic resources, we decided to incubate clutches in polyurethane hermetic boxes with constant monitoring and protection, as an alternative option to ensure hatchling recruitment.

Effects of artificial incubation

Physical handling of eggs can reduce hatching and emergence success in relocated clutches (Eckert and Eckert 1990; Pintus et al. 2009). However, in the present study, we found no significant differences in hatching and emergence success between *in situ* and artificially incubated clutches, indicating that handling was correctly carried out and that incubation conditions in boxes was suitable for hawksbill embryos development, as reported also for other species (e.g. Whitmore and Dutton 1985).

Concerning hatchling fitness, our results suggest that artificially incubated hatchlings from Saona are smaller than hatchlings from other Caribbean natural nesting rookeries, Mona Island (SCL = 4.0 cm: van Dam and Diez 1998), Cuba (SCL = 4.0 cm: Moncada et al. 1999) and the Virgin Islands (SCL = 4.1 cm: Hillis and Phillips 1996). However, the weight of Saona hatchlings seems to be similar to those from other Caribbean rookeries (14.8 g, Mona Island and 14.7 g, Virgin Islands).

Nonetheless, the use of boxes for incubation entails the risk of bias in the sex-ratio in favour of males (Dutton et al. 1985; Whitmore and Dutton 1985). Mean temperatures of 21 artificially incubated clutches during the thermosensitive period were below the pivotal temperature reference value from Mona Island (29.6ºC), suggesting that those temperatures are likely to have resulted in a male-skewed clutch sex-ratio. Moreover, since we found a clear relationship between temperature and incubation duration recorded in Saona, the long incubation durations recorded are also indicative of a male-skewed sex-ratio. The low percentages of female hatchling production inferred from our clutch temperature data, calls for urgent changes in the conservation strategy adopted in Saona, since a lengthy lack of female production would prevent population growth.

Forward strategies

Due to the rampant egg take, *in situ* incubation on the beaches is not an acceptable management practice for a long term conservation of this population. Hence, the primary goals in the near future would be: (1) to increase female production in boxes; and (2) to start a programme of relocation of clutches to safer areas (Pritchard 1995; DeGregorio and Southwood Williard 2011) while progressively reducing incubation in boxes. For the first goal, boxes are being exposed to direct sunlight during a few hours a day to raise incubation temperatures and increase the proportion of females produced (see Chantrapornsyl 1992). For the second goal, financial resources must be invested to strengthen the protection of a beach section near Mano Juan village as a safe area for clutch relocation.

Further research on beach temperature patterns is needed to increase understanding of the natural sex-ratio at Saona in order to improve conservation measures. Together with the artificial incubation programme we carried out additional measures such as hiring people from the local community to work as field technicians, organizing workshops and meetings with authorities, as well as environmental education and awareness programmes. These measures have apparently positively influenced local attitudes and perceptions toward marine turtles in Saona (White et al. 2011) which we strongly believe will favour a future programme of clutch relocation on the beach.

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REFERENCES

Almeida AP and Mendes SL (2007) An analysis of the role of local fishermen in the conservation of the loggerhead turtle (*Caretta caretta*) in Pontal do Ipiranga, Linhares, ES, Brazil. Biological Conservation, 134, 106 - 112.

Beggs JA, Horrocks JA and Krueger BH (2007) Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies. Endangered Species Research, 3, 159 - 168.

Carrillo E, Webb G and Manolis C (1999) Hawksbill turtles (*Eretmochelys imbricata*) in Cuba: an assessment of the historical harvest and its impacts. Chelonian Conservation and Biology, 3, 264 - 280.

Chan EH (1989) White spot development, incubation and hatching success of leatherback turtle (*Dermochelys coriacea*) eggs from Rantau Abang, Malaysia. Copeia, 1, 42 - 47.

Chantrapornsyl S (1992) Artificial incubation and embryonic development of olive ridley turtle eggs. Phuket Marine Biological Center Research Bulletin, 57, 41 - 50.

DeGregorio BA and Southwood Williard A (2011) Incubation temperatures and metabolic heating of relocated and in situ loggerhead sea turtle (*Caretta caretta*) nests at a northern rookery. Chelonian Conservation and Biology, 10, 54 - 61.

Dutton DL, Dutton PH, Chaloupka M and Boulon RH (2005) Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biological Conservation, 126, 186 - 194.

Dutton PH, Whitmore CP and Mrosovsky N (1985) Masculinisation of leatherback turtle *Dermochelys coriacea* hatchlings from eggs incubated in Styrofoam boxes. Biological Conservation, 31, 249 - 264.

Eckert KL (1999) Designing a conservation program. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ SSC Marine Turtle Specialist Group Publication No. 4, pp 6 - 8.

Eckert KL and Eckert SA (1990) Embryo mortality and hatch success in in situ and translocated leatherback sea turtle *Dermochelys coriacea* eggs. Biological Conservation, 53, 37 - 46.

Fleming EH (2001) Swimming against the tide: recent surveys of exploitation, trade, and management of marine turtles in the northern Caribbean. Washington, DC: TRAFFIC North America.

García A, Ceballos G and Adaya R (2003) Intensive beach management as an improved sea turtle conservation strategy in Mexico. Biological Conservation, 111, 253 - 261.

Godfrey MH and Mrosovsky N (1999) Estimating hatchling sex ratios. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 136 - 138.

Hawkes LA, Tomás J, Revuelta O, León YM, Blumenthal JM, Broderick AC, Fish M, Raga JA, Witt MJ and Godley BJ (2012) Migratory patterns in hawksbill turtles described by satellite tracking. Marine Ecology Progress Series, 461, 223 - 232.

Hillis ZM and Phillips B (1996) The year of expanding, standardizing and planning for the future of hawksbill turtle research at Buck Island reef NM. In: Proceedings of the 15th Annual Symposium on Sea Turtle Biology and Conservation, 20-25 February 1995. Hilton Head, South Carolina, USA.

Horrocks JA and Scott N McA (1991) Nest site location and nest success in the hawksbill turtle *Eretmochelys imbricata* in Barbados, West Indies. Marine Ecology Progress Series, 69, 1 - 8.

IUCN (2011) Red list of threatened species. Available at URL: http://www.iucnredlist.org (accessed 19 November 2011).

Kamel SJ and Delcroix E (2009) Nesting ecology of the hawksbill turtle, *Eretmochelys imbricata*, in Guadeloupe, French West Indies from 2000-07. Journal of Herpetology, 43, 367 - 376.

Kamel SJ and Mrosovsky N (2006) Deforestation: risk of sex ratio distortion in hawksbill sea turtles. Ecological Applications, 16, 923 - 931.

Kornaraki E, Matossiana DA, Mazarisa AD, Matsinosa YG and Margaritoulis D (2006) Effectiveness of different conservation measures for loggerhead sea turtle (*Caretta caretta*) nests at Zakynthos Island, Greece. Biological Conservation, 130, 324 - 330.

Lagueux CJ and Campbell CL (2005) Marine turtle nesting and conservation needs on the south-east coast of Nicaragua. Oryx, 39, 398 - 405.

Lagueux CJ, Campbell CL and McCoy WA (2003) Nesting and conservation of the hawksbill turtle, *Eretmochelys imbricata*, in the Pearl Cays, Nicaragua. Chelonian Conservation and Biology, 4, 588 - 602.

Liles MJ, Jandres MV, López WA, Mariona GI, Hasbún CR and Seminoff JA (2011) Hawksbill turtles *Eretmochelys imbricata* in El Salvador: nesting distribution and mortality at the largest remaining nesting aggregation in the eastern Pacific Ocean. Endangered Species Research, 14, 23 - 30.

Marcovaldi MA, Lopez G, Soaresi LS, Santos A, Bellini C and Barata P (2007) Fifteen years of hawksbill sea turtle (*Eretmochelys imbricata*) nesting in northern Brazil. Chelonian Conservation and Biology, 6, 223 - 228.

Martin RE (1996) Storm impacts on loggerhead turtle reproductive success. Marine Turtle Newsletter, 73, 10 - 12.

McClenachan L, Jackson JBC and Newman MJH (2006) Conservation implications of historic sea turtle nesting beach loss. Frontiers in Ecology and the Environment, 4, 290 - 296.

Meylan AB (1999) Status of the hawksbill turtle, *Eretmochelys imbricata*, in the Caribbean region. Chelonian Conservation and Biology, 3, 177 - 184.

Meylan AB (2002) Global Status Review of the hawksbill turtle (*Eretmochelys imbricata*), with an emphasis on the wider Caribbean. An Addendum for the Second CITES wider Caribbean hawksbill turtle dialogue meeting, Cayman Islands (U.K.) 21-23 May 2002. In Marine Turtle Specialist Group 2007 IUCN red list status assessment.

Miller JD (1999) Determining clutch size and hatch success. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 124 - 129.

Moncada F, Carrillo E, Saenz A and Nodarse G (1999) Reproduction and nesting of the hawksbill turtle, *Eretmochelys imbricata*, in the Cuban Archipelago. Chelonian Conservation and Biology, 3, 257 - 263.

Morreale SJ, Ruiz GJ, Spotila JR and Standora EA (1982) Temperature-dependent sex determination: current practices threaten conservation of sea turtles. Science, 216, 1245 - 1247.

Mortimer JA (1999) Reducing threats to eggs and hatchlings: hatcheries. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 175 - 178.

Mrosovsky N (1994) Sex ratios of sea turtles. Journal of Experimental Zoology, 270, 16 - 27.

Mrosovsky N and Pieau C (1991) Transitional range of temperature, pivotal temperatures and thermosensitive stages for sex determination in reptiles. Amphibia-Reptilia, 12, 169 - 179.

Mrosovsky N, Kamel SJ, Diez CE and van Dam RP (2009) Methods of estimating natural sex ratios of sea turtles from incubation temperatures and laboratory data. Endangered Species Research, 8, 147 - 155.

Öz M, Erdogan A, Kaska Y, Düsen S, Aslan A, Sert H, Yavuz M and Tunc MR (2004) Nest temperatures and sex-ratio estimates of loggerhead turtles at Patara beach on the south western coast of Turkey. Canadian Journal of Zoology, 82, 94 - 101.

Pérez-Castañeda R, Salum-Fares A and Defeo O (2007) Reproductive patterns of the hawksbill turtle *Eretmochelys imbricata* in sandy beaches of the Yucatan Peninsula. Journal of the Marine Biological Association of the United Kingdom, 87, 815 - 824.

Pintus KJ, Godley BJ, McGowan A and Broderick AC (2009) Impact of clutch relocation on green turtle offspring. Journal of Wildlife Management, 73, 1151 - 1157.

Pritchard P (1995) Recovered sea turtle populations and U.S. recovery team efforts. In: Bjorndal K (ed). Biology and conservation of sea turtles. Washington, DC: Smithsonian Institution Press, pp 503 - 511.

Revuelta O, León YM, Feliz P, Godley BJ, Raga JA and Tomás J (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. Oryx, 46, 348 - 358.

Richardson JI, Hall DB, Mason PA, Andrews KM, Bjorkland R, Cai Y and Bell R (2006) Eighteen years of saturation tagging data reveal a significant increase in nesting hawksbill sea turtles (*Eretmochelys imbricata*) on Long Island, Antigua. Animal Conservation, 9, 302 - 307.

Sarti Martínez L, Barragán AR, García Muñoz D, García N, Huerta P and Vargas F (2007) Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology, 6, 70 - 78.

Sato F and Madriasau BB (1991) Preliminary report on natural reproduction of hawksbill sea turtles in Palau. Marine Turtle Newsletter, 55, 12 - 14.

Sokal RR and Rohlf FJ (1995) Biometry: the principles and practice of statistics in biological research. 3rd edition. New York: W.H. Freeman and Co.

van Dam RP and Diez CE (1998) Caribbean hawksbill turtle morphometrics. Bulletin of Marine Science, 62, 145 - 155.

Wallace BP, Dimatteo AD, Hurley BJ, Finkbeiner EM, Bolten AB, Chaloupka MY, Hutchinson BJ, Abreu-Grobois FA, Amorocho D, Bjorndal KA, Bourjea J, Bowen BW, Briseño Dueñas R, Casale P, Choudhury BC, Costa A, Dutton PH, Fallabrino A, Girard A, Girondot M, Godfrey MH, Hamann M, López-Mendilaharsu M, Marcovaldi MA, Mortimer JA, Musick JA, Nel R, Pilcher NJ, Seminoff JA, Troëng S, Whiterington B and Mast RB (2010) Regional management units for marine turtles: framework for prioritizing conservation and research across multiple scales. PloS One. doi:10.1371/journal.pone.0015465.

White TH Jr, Jhonson Camacho A, Bloom T, Lancho Diéguez P and Sellares R (2011) Human attitudes and perceptions regarding endangered species conservation: a case study of

Saona Island, Dominican Republic. Latin American Journal of Conservation, 2, 18 - 29.

Whitmore CP and Dutton PH (1985) Infertility, embryonic mortality and nest-site selection in leatherback and green sea turtles in Suriname. Biological Conservation, 34, 251 - 272.

Wibbels T (2003) Critical approaches to sex determination in sea turtles. In: Lutz PL, Musick JA and Wyneken J (eds). The biology of sea turtles. Volume II. Boca Raton, FL: CRC Press, pp 103 - 134.

Yntema CL and Mrosovsky N (1982) Critical periods and pivotal temperatures for sexual differentiation in loggerhead turtles. Canadian Journal of Zoology, 60, 1012 - 1016.

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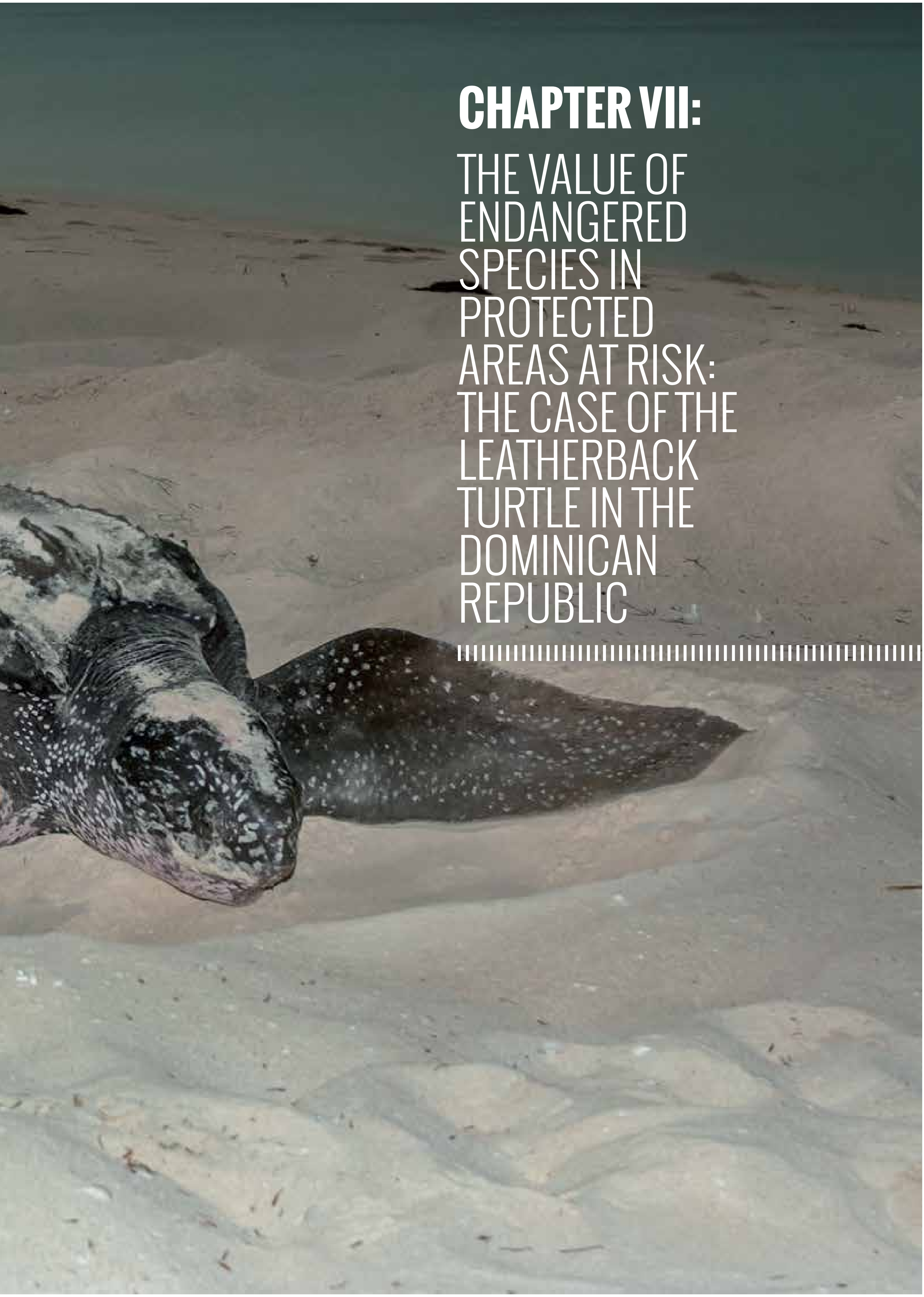

CHAPTER VII:

THE VALUE OF ENDANGERED SPECIES IN PROTECTED AREAS AT RISK: THE CASE OF THE LEATHERBACK TURTLE IN THE DOMINICAN REPUBLIC

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THE VALUE OF ENDANGERED SPECIES IN PROTECTED AREAS AT RISK: THE CASE OF THE LEATHERBACK TURTLE IN THE DOMINICAN REPUBLIC.

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ABSTRACT

Protected areas are considered essential elements for global biodiversity conservation. They may not necessarily result in an effective conservation of resources in developing countries due to lack of funding for management and enforcement. In addition, poor governance aligned with conflicts of economic interests related to their use can further threaten their integrity and persistence. In the Dominican Republic, the western beaches of the Jaragua National Park, a protected area which is also part of a UNESCO Biosphere Reserve, have been proposed for development using a mass-tourism model. One of the most charismatic species found in this area is the leatherback turtle (*Dermochelys coriacea*). In the present study, we assess hatching success, and factors affecting it, to determine the reproductive value across the area for the leatherback turtle. The main factors found driving hatching success at the study beaches are beach sector, incubation duration, date of lay and clutch size. Our results show that clutches in La Cueva (located in the buffer zone of the park) and Bahía de las Águilas (located inside the limits of the park) have an unusually high hatching success (~75%) for this species, highlighting the importance of increasing protection efforts at these sites. We strongly recommend including La Cueva inside the limits of the Jaragua National Park.

INTRODUCTION

Protected areas remain a cornerstone of global conservation efforts to preserve diminishing wildlife species and their habitats (CBD 2010; Butchart et al. 2012). In the Caribbean, the establishment of Marine Protected Areas (MPAs) to protect natural resources, unique habitats and threatened species have proliferated over the last few decades (Guarderas et al. 2008). Nevertheless, in many countries not all protected areas have management plans, and when they exist, the national authorities responsible for their protection are often under-resourced, making protection less effective (e.g. Ervin 2003; Buitrago et al. 2008).

The Dominican Republic (DR) is a developing country experiencing a rapid increase in international tourism, particularly focussed on coastal areas of the north and southeast regions (León 2007; Wielgus et al. 2010). In recent years, this mass-tourism model and the expansion of existing bauxite and limestone mining have been proposed as valuable ways to enhance the impoverished economy of the southwest of the country (Wielgus et al. 2010). It is known that these activities could result in an adverse impact on natural environments such as pollution and sand mining, with detrimental effects on reefs and seagrass ecosystems (Geraldes 2003; Grandoit 2005).

At the southwest of the country, the Bahía de las Águilas hosts one of the last refuges of coastal marine fauna and flora available to the Dominican Republic. The ecosystems are thought unlikely to be able to support mass tourism (Wielgus et al. 2010). This area has unique vegetation with several endemic plants and, among its marine ecosystems; there are the most extensive and best preserved seagrass beds in the country. Coral reefs are found a short distance from the coast, so it is highly probable that these reefs would be greatly affected by any land-based pollution, particularly resultant from mining activity. Finally, the bay hosts the reproductive areas of rhinoceros iguana (*Cyclura cornuta*) and marine turtles (Rupp et al. 2005; Revuelta et al. 2012).

One of the most charismatic species in this area is the leatherback turtle (*Dermochelys coriacea*). Although populations of this species seem to be recovering in the North Atlantic, and it has assigned to be under “low risk a low threat” (Wallace et al. 2011), in some rookeries this species face serious threats, such as egg take that compromise their conservation (e.g., Patiño-Martínez et al. 2008). Moreover, the leatherback turtle exhibits what is considered a very low hatch success rate (~50%) in comparison to other marine turtle species (Bell et al. 2003; references in Eckert et al. 2012).

In the past, the leatherback turtle was reported nesting widely around the coast of the DR (Ottenwalder 1981), but today the last important nesting areas for the species are located in the beaches of the Jaragua National Park (JNP; Revuelta et al. 2012). There are two leatherback nesting areas in the Park, the beaches adjacent to the Oviedo lagoon at the east, which experienced 100% egg take by local people, and Bahía de las Águilas and La Cueva beaches in the west of the Park (Revuelta et al. 2012), but La Cueva is only currently within the buffer zone of the Park (see study site section). Since 2006, a cooperative marine turtle conservation project has been carried out at the JNP and, as a result, leatherback clutches are currently monitored for protection by rangers at the western nesting beaches. However, the strong pressure to bring mass tourism to these beaches in the form of high capacity resorts represents a potential threat to protected fauna of the Park (Rupp et al. 2005) including nesting marine turtles. The JNP was created in 1983 and since 2002 it has been part of the UNESCO Jaragua-Bahoruco-Enriquillo Biosphere Reserve. Multiple national laws as well as International agreements ratify protection for the Bahía de las Águilas territory and/or its natural resources: The General Environmental Law 64-00, the Sectorial Law for Protected Areas 202-04, The Convention on Biodiversity, and The Cartagena Protocol for Wildlife Areas with Special Protection. In Bahía de las Águilas, building and extraction of animals or plants are prohibited, but visits of tourists by boat or by 4x4 vehicles into the beach are permitted. A ranger post is placed at the entrance of the bay, and rangers patrol the beach regularly. In the buffer zone of the park there is no patrolling and access to the beach is uncontrolled.

Here we set out (1) to study hatching success and factors affecting it to determine the reproductive value across the area for the leatherback turtle and, (2) to ascertain the importance of both nesting beaches for this threatened species to reinforce conservation in the face of anthropogenic threats at site.

MATERIALS AND METHODS

Study site

The present study was conducted between 2007 and 2010 at the western beaches of The Jaragua National Park (JNP, N17° 57', W71° 39') a protected area of 1374 km^2 (of which 905 km^2 is marine reserve) situated in the south-western corner of the DR (Figure 7.1). Bahía de las Águilas is a 4.4 km sandy bay located inside the park but is also considered as a national recreational area according to local laws and is frequented by several thousand Dominican and international tourists per annum. La Cueva beach (2.5 km in length) is located in the northern buffer zone of the park. A small village (around 20 inhabitants) and a rangers' post are located between these two beaches that are separated by a rocky zone of 1 km length.

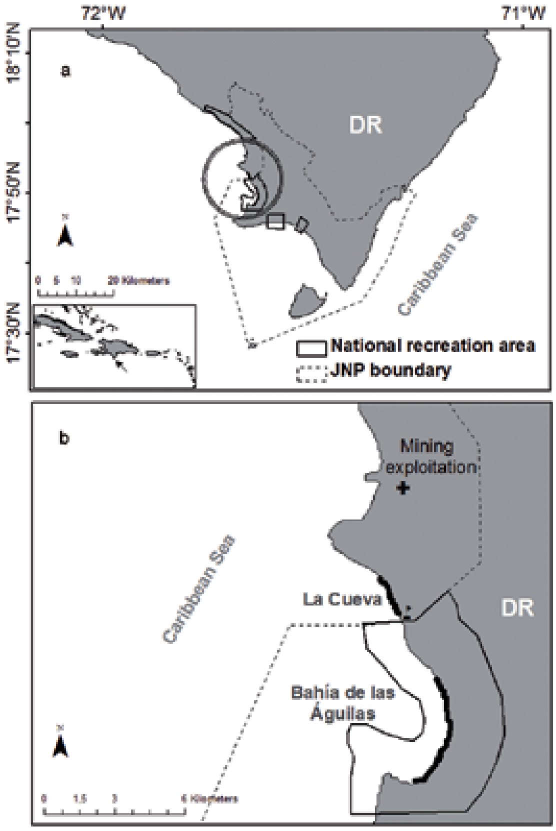


Figure 7.1. a) Map of the Jaragua National Park (SW DR) showing the park limits and the areas of recreational use inside it. The insets indicate the location of the main maps in the Caribbean. The circle indicates the area enlarged in b. b) Leatherback nesting beaches at the West of the Park, Bahía de las Águilas and La Cueva.

Data collection

During the leatherback nesting season (from March to August, with a peak in April-June), we carried out night and daytime surveys at Bahía de las Águilas and La Cueva nesting beaches (See Revuelta et al. 2012 for an exhaustive description). When we found a clutch, we camouflaged the tracks of females and recorded GPS coordinates; we also recorded distance from nest to high tide line, the beach sector (Bahía de las Águilas and La Cueva), and the zone of the beach where the clutch was located (open sand, or within the dune vegetation).

We studied a total of 109 clutches laid on the beaches of Bahía de las Águilas and La Cueva over four nesting seasons (2007-2010). We excavated clutches by hand one to three days after hatchling emergence was detected. Clutch size was defined as the total number of eggs (hatched and unhatched yolked eggs) ·clutch^{-1}. Hatching success was calculated by dividing the number of eggs hatched by the clutch size expressed as percentage (Miller 1999). Leatherback turtle clutches have a high frequency of eggs without yolk commonly referred to as yolkless eggs which were also counted during the excavation but are not accounted for in clutch size.

Statistical analysis

We undertook detailed data exploration before any statistical analysis following Zuur et al. (2010). When the underlying question is to determine which covariates are driving a system, the most difficult aspect of the analysis is probably how to deal with correlation between covariates. We used variance inflation factor (VIF) and multi-panel scatterplots to test for covariate collinearity. We used boxplot and Cleveland plots for outlier detection, and two-way relationships were studied through multi-panel scatterplots between the response variable (percent hatching success) and each covariate.

Restricted maximum likelihood with linear mixed models was used to investigate the effect of environmental and temporal variables on the hatching success. The response variable (percent hatching success) was logit transformed and analysed as a function of the explanatory variables considered likely to be important determinant predictors of the hatching success for sea turtles in previous studies (Wallace et al. 2004; Ditmer and Stapleton 2012; See Table 7.1 for description). We included year as random factor in our model. It should be noted that we excluded 2010 data from statistical analyses because some of the predictor variables were not recorded in this year due to fieldwork limitations. To identify a suitable and parsimonious approximating model, we first developed a series of alternative mixed effects models that included different combinations of the explanatory variables using stepwise process. Model selection was based on ΔAIC values lower than 2, calculated as the difference between the AIC values for each model and the model with lowest AIC, and model weights (wi) (Burnham and Anderson 2002). Model weight is a relative index for model's likelihood against any other model in the set and it was also used to calculate the relative importance of a variable by summing the weights of all the models that include that variable (Burnham and Anderson 2002).

All analyses were performed using the lnme package (Pinheiro et al. 2011) in R (version 2.14.0) for the linear mixed models.

Table 7.1. List of explanatory variables included in mixed models to model the leatherback hatching success in the JNP. The name of each variable entered into the models, description of each one, type and levels of the categorical variables are presented.

Name	Variable description	Type of variable	Levels of variables
LOCATION			
DIST	Distance (m) to high tide line	Continuous	
ZONE	Beach zone	Categorical	Open sand/Vegetation
SECT	2 sections separated by a rocky zone	Categorical	Bahía de las Águilas/ La Cueva
TEMPORAL			
YEAR	Breeding season year	Categorical	2007, 2008, 2009
JULIAN	Date of clutch laying	Continuous	
REPRODUCTIVE			
ClutchSZ	Clutch size (number of yolked eggs)	Continuous	
YLS	Number of yolkless eggs	Continuous	
ID	Incubation duration (number of days between egg laying and the time of first hatchling emergence)	Continuous	

RESULTS

In Bahía de las Águilas beach, mean ± SD hatching success of leatherback turtles clutches across years was 54.9 ± 15.5% (n = 6), 83.5 ± 20.9% (n = 19), 66.1 ± 24.9% (n = 16) and 76.1 ± 20.3% (n = 40) in 2007-2010, respectively). In La Cueva beach, mean ± SD hatching success was 81.5 ± 16.3 (n = 6) in 2007, 79.1 ± 12.0 (n = 17) in 2009 and 84.2 ± 9.6 (n = 5) in 2010 (Figure 7.2a). No clutches were studied in 2008 in La Cueva beach.

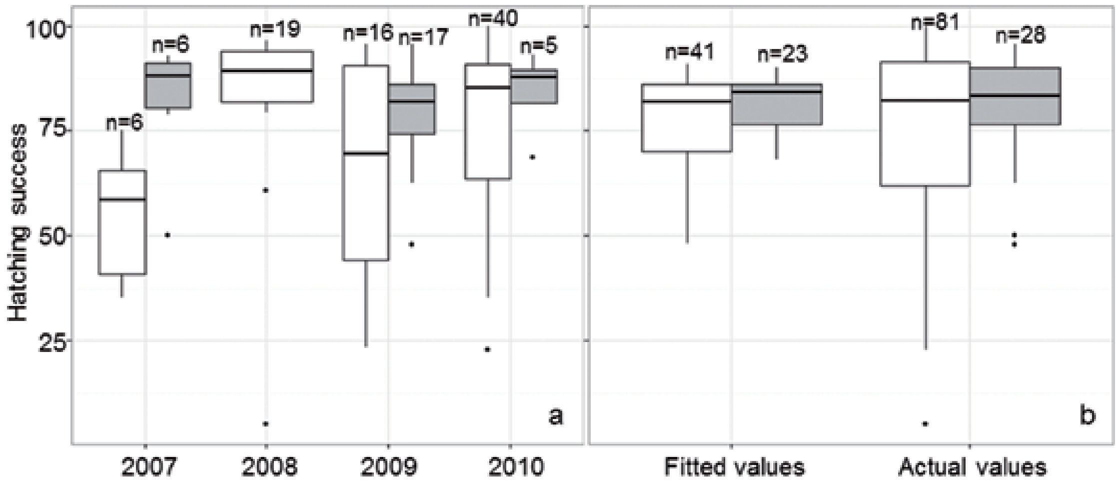


Figure 7.2. a) Hatching success of leatherback turtles in Bahía de las Águilas (white) and La Cueva beach (grey). a) Actual data across years during the study period. No clutches were incubated in La Cueva in 2008. b) Fitted values by locality predicted by the best model (see Table 7.2) compared with actual values. Numbers above bars indicate sample size.

We analyzed and modeled the hatching success of 64 clutches laid by leatherback turtles spanning three nesting seasons (2007, 2008 and 2009). No correlation was found between any pair of variables which were, therefore, included in the model testing the effect of 7 biotic and abiotic covariates (see Table 7.1). Of the 25 candidate models considered, five were retained in the models selected by AIC (Table 7.2). The hatching success of leatherback turtles appeared to be mainly driven by the effects of characteristics of beach sector, the incubation duration and date of lay, as evidenced by the retention of these variables in the most parsimonious model (AIC= 211.32) and in all five models with ΔAIC values lower than 2 (Table 7.2). The relative importance of beach sector (Bahía de las Águilas and La Cueva), incubation duration and date of lay was 0.99 and the relative importance of clutch size, beach zone (open sand, or within the dune vegetation) and distance to high tide line, was 0.75, 0.35 and 0.24, respectively. Although the number of yolkless eggs was initially included in the models, this variable was not significant.

According to the best model, hatching success of clutches incubated in the beach sector of La Cueva was higher than in those incubated in Bahía de las Águilas (Table 7.3, Figure 7.2b).

Table 7.2. List of models with better fit to the data are presented sorted according to Akaike information criterion (AIC) values. Five (out of 25) models with values of ΔAIC lower than 2 are presented. Weights of evidence in support of a particular model given the data (*w*) are also listed.

Models	AIC	ΔAIC	w
HS ~ SECT + ID + JULIAN + ClutchSZ + YEAR (random factor)	211.32	0.00	0.28
HS ~ SECT + ID + JULIAN + ClutchSZ + DIST + YEAR (random factor)	211.60	0.28	0.24
HS ~ SECT + ID + JULIAN + ClutchSZ + ZONE + YEAR (random factor)	211.78	0.46	0.23
HS ~ SECT + ID + JULIAN +YEAR (random factor)	212.98	1.66	0.12
HS ~ SECT + ID + JULIAN + ZONE +YEAR (random factor)	213.12	1.79	0.12

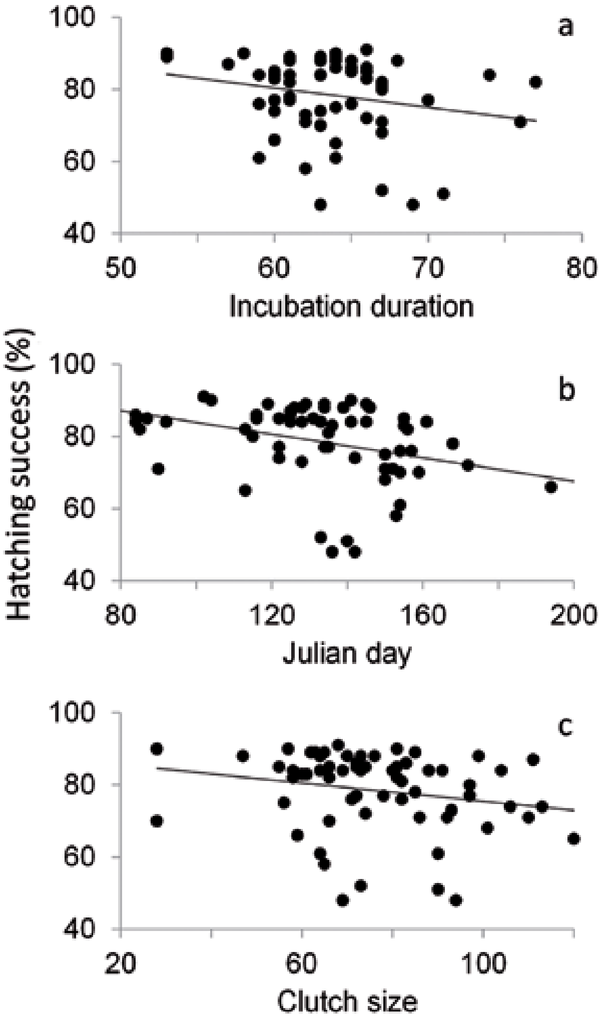
Table 7.3. Estimates and standard errors (SE) from the selected model to explain the factors affecting hatching success of leatherback turtles in JNP. Model terms are described in Table 7.1.

Variable	Estimate	SE
Intercept	10.227***	2.903
SECTOR La Cueva	0.711**	0.329
ID	-0.088**	0.036
JULIAN	-0.016**	0.006
ClutchSZ	-0.017*	0.009

Significance codes: *** P = 0-0.001; ** P = 0.01-0.05; * P = 0.05-0.1

In addition to beach sector, hatching success was strongly influenced by incubation duration (Table 7.3); longer incubation durations resulted in lower hatching success (Figure 7.3a). We also found date of lay affecting hatching success, with clutches laid earlier in the nesting season having higher hatching success (Table 7.3, Figure 7.3b). The model also showed that increasing clutch size had a negative effect on hatching success (Table 7.3, Figure 7.3c).

Figure 7.3. Effect of a) incubation duration, b) julian day (date of lay) and c) clutch size (number of eggs incubated), on hatching success of clutches at western beaches of JNP. Shown are fitted values of the best model. Trends lines are cubic smoothing splines fitted by generalized cross-validation.



DISCUSSION

Our results show that clutches of leatherback turtles in the western beaches of the JNP presented unusually high hatching success (75.2%) for this species in the Caribbean (~ 50% see Eckert et al. 2012 for review). These beaches are of high value for the recovery of this threatened species in the Dominican Republic, taking into account the high level of egg take and the difficulties in protection in other beaches in the country (Revuelta et al. 2012). Studies on other beaches in the Caribbean region have suggested that bacterial and fungal infections (Patiño-Martínez et al. 2011), egg predation by ghost crabs, ants and other arthropoda (Maros et al. 2003; Caut et al. 2006) or non-natural predators such as feral dogs (Ordoñez et al. 2007) are important factors reducing hatching success. Additionally, the invasion of the clutch by vegetation roots, particularly from the species *Ipomoea pes-caprae*, may also have adverse effects on hatching success (Caut et al. 2010; Conrad et al. 2011). Although no quantitative data were recorded, fungal attacks (according to description in Chan and Solomon 1989) were detected in a very few clutches. The ghost crab (*Ocypode quadrata*) is present at the beaches; however, we did not observe egg predation by crabs, nor did we observe predation by ants or dogs or inundation by roots. Moreover, *Ipomea pes-caprae* is not present in the western beaches of the JNP. Taken together the absence of these factors likely contributes to enhanced hatch rate at site. Hence, our results highlight the need to preserve the relatively pristine environments with native beach vegetation to maintain the current levels of hatchling production.

Although both sectors studied presented similar nest density, sand and vegetation, clutches at La Cueva beach had even higher hatching success than at Bahía de las Águilas, possibly due to a greater slope in the former. Apart from beach sector, the study of factors affecting hatching success revealed that incubation duration and date of lay affected hatching success. The lower hatching success found in those clutches with longer incubation durations might be associated with process of washover which could cause temperature decrease inside the clutch, thus affecting embryo development (Houghton et al. 2007; Caut et al. 2010). Furthermore, the Caribbean hurricane season starts in June when many clutches are still incubating on the beach. Hence, it is possible that the low hatching success found in clutches laid later in the season might be associated with variations on incubation temperatures due to more violent wave regimes or flooding (Santidrián Tomillo et al. 2009). Future research should explore the roles of other factors that could not be recorded in this study such as nest depth, sand structure and incubation temperature, as well as individual-based reproductive variables that significantly affected hatching success in other nesting areas (Rafferty et al. 2011; Perrault et al. 2012). Since no spatial factors such as beach zone or distance to the shoreline seemed to affect hatching success, *in situ* clutch protection seems to be sufficient to preserve hatchling production in these beaches, avoiding the potentially negative effects of clutch relocation on hatching success (Pintus et al. 2009).

Current management strategies include surveillance of beaches and clutches by the park rangers in Bahía de las Águilas. However, the Dominican government lacks the necessary human and economic resources to effectively manage protected areas and offenses to environmental laws are frequent in these beaches, as happens in other protected areas of the country (García and Roersch 1996; Kerchner et al. 2010). Given the exceptional value of hatching success in these beaches and current and potential threats affecting leatherback nesting beaches in the country, additional effort in regulation and management of this protected area is needed. Potential actions that can be taken to improve management strategies should include increased beach surveillance by rangers during the leatherback nesting season, banning access of motorized vehicles to the beaches as well as the use of lights and campfires. Government policies should include increasing awareness to facilitate a reduction in pollution, preserve the environment, and protect the endangered species of the Park (Choi and Eckert 2009).

An added threat to the lack of enforcement is the strong pressure to bring mass tourism to this area (both in the protected and non-protected beaches) in the form of high capacity resorts. Our study is particularly relevant in relation to La Cueva beach; this sector harbors 20% of the total clutches laid at western beaches of the Park and demonstrated the highest hatching success. However, it is less protected because it is located outside the Park limits, in the buffer zone. This means a higher level of threat as has been observed elsewhere in the country (Kerchner et al. 2010). Allowing coastal development in the buffer zone would increase detrimental effects of human activities that would impact no only this zone but also the areas inside the Park. We strongly recommend including this beach inside the limits of the park thus conferring the same level of legal protection and surveillance as Bahía de las Águilas.

CONCLUSIONS

Although Bahía de las Águilas is one of the best preserved areas in the Dominican Republic, hosting endemic species, it is threatened with increasing interest in development with plans including building of resorts and tourist facilities. Our study highlights the importance of this site and the neighbouring beach of La Cueva for leatherback turtle reproduction and the need of establishing management measures for a successful *in situ* conservation of this threatened species in the country. These protection measures would be also beneficial for the preservation of other species.

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REFERENCES

Bell BA, Spotila JR, Paladino FV and Reina RD (2003) Low reproductive success of leatherback turtles, *Dermochelys coriacea*, is due to high embryonic mortality. Biological Conservation, 115, 131 - 138.

Buitrago J, Guada HJ and Doyle E (2008) Conservation science in developing countries: an inside perspective on the struggles in sea turtle research and conservation in Venezuela. Environmental Science and Policy, 11, 562 - 578.

Burnham K and Anderson D (2002) Models selection and multi-model inference. Springer Verlag, Berlin.

Butchart SHM, Scharlemann PW, Evans MI, Quader S, Arico S, Arinaitwe J, Balman M, Bennun LA, Bertzky B, Besancon C, Boucher TM, Brooks TM, Burfield IJ, Burgess ND, Chan S, Clay RP, Crosby MJ, Davidson NC, De Silva N, Devenish C, Dutson GCL, Díaz Fernández DF, Fishpoo LDC, Fitzgerald C, Foster M, Heath MF, Hockings M, Hoffmann M, Knox D, Larsen FW, Lamoreux JF, Loucks C, May I, Millett J, Molloy D, Morling P, Parr M, Ricketts TH, Seddon N, Skolnik B, Stuart SN, Upgren A and Woodley S (2012) Protecting important sites for biodiversity contributes to meeting global conservation targets. PLoS ONE. doi:10.1371/journal.pone.0032529.

Caut S, Guirlet E, Jouquet P and Girondot M (2006) Influence of nest location and yolkless eggs on the hatching success of leatherback turtle clutches in French Guiana. Canadian Journal of Zoology, 84, 908 - 815.

Caut S, Guirlet E and Girondot M (2010) Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. Marine Environmental Research, 69, 254 - 261.

CBD (2010) Secretariat of the Convention on Biological Diversity. Global Biodiversity Outlook 3. Montreal, pp 1 - 94.

Chan EH and Solomon SE (1989) The structure and function of the eggshell of the leatherback turtle (*Dermochelys coriacea*) from Malaysia, with notes on attached fungal forms. Animal Technology, 40, 91 - 102.

Choi GY and Eckert KL (2009) Manual of best practices for safeguarding sea turtle nesting beaches. Wider Caribbean Sea Turtle Conservation Network (WIDECAST) Technical Report No. 9. Ballwin, Missouri. 86 pp.

Conrad JR, Wyneken J, Garner JA and Garner S (2011) Experimental study of dune vegetation impact and control on leatherback sea turtle *Dermochelys coriacea* nests. Endangered Species Research, 15, 13 - 27.

Ditmer MA and Stapleton SP (2012) Factors affecting hatch success of hawksbill sea turtles on Long Island, Antigua, West Indies. PLoSONE. doi:10.1371/journal.pone.0038472.

Eckert KL, Wallace BP, Frazier JG, Eckert SA and Pritchard PCH (2012) Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication BTP - R4015-2012, Washington, D.C.

Ervin J (2003) Protected area assessments in perspective. Bioscience, 53, 819 - 822.

García R and Roersch C (1996) Politica de manejo y utilización de los recursos florísticos en la República Dominicana. Journal of Ethnopharmacology, 51, 147 - 160.

Geraldes FX (2003) The coral reefs of the Dominican Republic. In Cortés J (ed) Latin American Coral Reefs. Elsevier, Amsterdam, Holanda, pp 303 - 330.

Grandoit J (2005) Tourism as a development tool in the Caribbean and the environmental by-products: the stresses on small island resources and viable Remedies. Journal of development and social transformation 2. Available from http://www.maxwell.syr.edu/moynihan/programs/dev/journal.htlm (accessed January 19, 2013).

Guarderas AP, Hacker SD and Lubchenco J (2008) Current status of marine protected areas in Latin America and the Caribbean. Conservation Biology, 22, 1630 - 1640.

Houghton JDR, Myers AE, Lloyd C, King RS, Isaacs C and Hays GC (2007) Protracted rainfall decreases temperature within leatherback turtle (*Dermochelys coriacea*) clutches in Grenada, West Indies: Ecological implications for a species displaying temperature dependent sex determination. Journal of Experimental Marine Biology and Ecology, 345, 71 - 77.

Kerchner C, Honzák M, Kemkes R, Richardson A, Townsend J and Rimmer CC (2010) Designing spatially explicit incentive programs for habitat conservation: A case study of the Bicknell's thrush wintering grounds. Ecological Economics, 69, 2108 - 2115.

León YM (2007) The impact of tourism on rural livelihoods in the Dominican Republic's coastal areas. Journal of Development Studies, 43, 340 - 359.

Maros A, Louveaux A, Godfrey MH and Girondot M (2003) *Scapteriscus didactylus* (Orthoptera, Gryllotalpidae), predator of leatherback turtle eggs in French Guiana. Marine Ecology Progress Series, 249, 289 - 296.

Miller JD (1999) Determining clutch size and hatch success. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of seaturtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 6 - 8.

Ordoñez C, Troëng S, Meylan A, Meylan P and Ruiz A (2007) Chiriqui Beach, Panama, the most important leatherback nesting beach in Central America. Chelonian Conservation and Biology, 6, 122 - 126.

Ottenwalder JA (1981) Estudio preliminar sobre el estado, distribución, y biología reproductiva de las tortugas marinas en la República Dominicana. Tesis de Licenciatura, Universidad Autónoma de Santo Domingo, Dominican Republic.

Patiño-Martínez J, Marco A, Quiñones L and Godley B (2008) Globally significant nesting of the leatherback turtle (*Dermochelys coriacea*) on the Caribbean coast of Colombia and Panama. Biological Conservation, 141, 1982 - 1988.

Patiño-Martínez J, Marco A, Quiñones L, Abella E, Muriel Abad R and Diéguez-Uribeondo J (2011) How do hatcheries influence embryonic development of sea turtle eggs? Experimental analysis and isolation of microorganisms in leatherback turtle eggs. Journal of Experimental Zoology, 313A.

Perrault JR, Miller DL, Eads E, Johnson C, Merrill A, Thompson LJ and Wyneken J (2012) Maternal health status correlates with success of leatherback sea turtles (*Dermochelys coriacea*) from Florida. PLoSONE. doi:10.1371/journal.pone.0031841.

Pinheiro J, Bates D, DebRoy S and Sarkar D, R Development Core Team (2011) nlme: Linear and nonlinear mixed effects models. R package version 3.1 - 102.

Pintus KJ, Godley BJ, McGowan A and Broderick AC (2009) Impact of clutch relocation on green turtle offspring. Journal of Wildlife Management, 73, 1151 - 1157.

Rafferty AR, Santidrián Tomillo P, Spotila JR, Paladino FV and Reina RD (2011) Embryonic death is linked to maternal identity in the leatherback turtle (*Dermochelys coriacea*). PLoS ONE. doi:10.1371/journal.pone.0021038.

Revuelta O, León YM, Feliz P, Godley BJ, Raga JA and Tomás J (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. Oryx, 46, 348 - 358.

Rupp E, Incháustegui SJ and Arias Y (2005) Conservation of *Cyclura ricordii* in the southwestern Dominican Republic and a brief history of the Grupo Jaragua. Iguana, 12, 222 - 234.

Santidrián Tomillo P, Suss JS, Wallace BP, Magrini KD, Blanco G, Paladino FV and Spotila JR (2009) Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology, 156, 2021 - 2031.

Wallace BP, Sotherland PR, Spotila JR, Reina RD, Franks BF and Paladino FV (2004) Biotic and abiotic factors affect the nest environment of embryonic leatherback turtles, *Dermochelys coriacea.* Physiological and Biochemical Zoology, 77, 423 - 432.

Wallace BP, DiMatteo AD, Bolten AB, Chaloupka MY, Hutchinson BJ et al (2011) Global Conservation Priorities for Marine Turtles. PLoS ONE. doi:10.1371/journal.pone.0024510.

Wielgus J, Cooper E, Torres R and Burke L (2010) Coastal Capital: Dominican Republic. Case studies on the economic value of coastal ecosystems in the Dominican Republic. Working paper. World Resources Institute, Washington, DC, USA.

Zuur AF, Leno EN and Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution, 1, 3 - 14.

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CHAPTER VIII:

EVALUATING THE IMPORTANCE OF MARINE PROTECTED AREAS FOR THE CONSERVATION OF HAWKSBILL TURTLES (*ERETMOCHELYS IMBRICATA*) NESTING IN THE DOMINICAN REPUBLIC

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CHAPTER VIII:

EVALUATING THE IMPORTANCE OF MARINE PROTECTED AREAS FOR THE CONSERVATION OF HAWKSBILL TURTLES (*ERETMOCHELYS IMBRICATA*) NESTING IN THE DOMINICAN REPUBLIC

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Endangered Species Research (Submitted)

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ABSTRACT

Understanding spatial and temporal habitat-use patterns to protect both foraging and breeding grounds of species of concern is crucial for successful conservation. Saona Island in Del Este National Park (DENP), southeastern Dominican Republic (DR), hosts the only major hawksbill (*Eretmochelys imbricata*) nesting area in the DR (100 nests.year^{-1} ± 8.4 s.d., range: 93-111), with the population having been critically reduced through hunting. We satellite tracked nine hawksbill turtles and present analyses of their core-use areas with respect to Marine Protected Areas both in their internesting and foraging areas. Kernel utilization distributions indicated that during the internesting period all turtles remained close to their nesting beaches in small home ranges in the territorial waters of DR, mostly over the continental shelf (<200 m depth). Common core-use areas were located inside the DENP and 82.7 % of all locations were within the DENP. At foraging areas, only 23 % of locations were inside MPAs either in waters of the DR and in waters of Bahamas, Nicaragua and Honduras. Our results highlight that the protected areas of the DR are key for hawksbill conservation, and enforcement of existing legislation of the protected areas in the country is key. The present study also corroborates that the waters off Nicaragua and Honduras are exceptionally important foraging areas for hawksbills in the Caribbean, showing the turtle's vulnerability in these waters.

INTRODUCTION

Many threatened marine vertebrate species are of conservation concern as the result of a range of past and ongoing threats (Read et al. 2006; Hoffman et al. 2011); and the establishment of marine protected areas (MPAs) has been promoted as a key management measure for their conservation (Halpern 2014). Management and conservation effectiveness of MPAs requires full knowledge about life history stages of the species they are intended to conserve (Edgar 2011). For highly migratory marine species this means knowledge about their migration routes as well as their movements and use of foraging and breeding grounds (Blanco et al. 2012; Costa et al. 2012). Owing to a revolution in the use of location technologies for tracking the at-sea movements of a range of marine species ('biologging', Bograd et al. 2010; Hart and Hyrenbach 2010; Hammerschlag et al. 2011), we now know a great deal about the movements and behaviour of many formerly cryptic marine species and arguably the most about the marine turtles (Godley et al. 2008; Maxwell et al. 2011; Scott et al. 2012; Schofield et al. 2013).

The hawksbill turtle (*Eretmochelys imbricata*) is the most tropical of all marine turtles, distributed in tropical and subtropical areas of the Atlantic, Pacific and Indian oceans, and currently is listed globally as critically endangered (IUCN 2013). In the Caribbean, this species is a major predator in coral reef ecosystems, with sponges constituting the main component of its diet (Meylan 1988; León and Diez 1999). Numerous studies have suggested the key role of this species in maintaining the structure, ecology and evolution of coral reefs (León and Bjorndal 2002; Bjorndal and Jackson 2003). During the nesting season, which usually spans several months, females are thought to return repeatedly to the same beach to lay a variable number of clutches, thus presumably remaining close to the nesting beach (Marcovaldi et al. 2012; Walcott et al. 2012). After breeding, hawksbill turtles are then thought to migrate away from the nesting beach to foraging grounds (Meylan et al. 2011). Migratory movements of this species in the Caribbean, tracked using satellite telemetry, show a broad range of movements, including turtles remaining in waters proximate to the nesting beaches and others migrating to foreign waters many hundreds to thousands of kilometres away (van Dam et al. 2008; Horrocks et al. 2011; Hawkes et al. 2012; Moncada et al. 2012).

The Dominican Republic (DR) in the Eastern Caribbean, hosts regionally significant numbers of nesting hawksbill turtles (Revuelta et al. 2012) but current nesting is largely restricted to protected areas. Saona Island in Del Este National Park (DENP), southeastern DR, hosts the last major nesting area in the country (~100 nests per year; Revuelta et al. 2012). Although nesting beaches are in a protected area, there has been a marked reduction in the abundance of this species and egg take is still a major threat at these beaches (Revuelta et al. 2012; 2013).

Here we reanalyse these tracking data with respect to (1) behaviour of hawksbill females during the internesting period and, (2) an assessment the level of protection afforded to these turtles in their internesting and foraging areas.

MATERIALS AND METHODS

Tagging area

Surveys for nesting hawksbill turtles were carried out in the Jaragua National Park (JNP, southwest DR, 17816′N-71533′W; Figure 8.1) and, Del Este National Park (DENP, southeastern DR, 18807′N-68844′W; Figure 8.1). Both parks were added to the UNESCO World Heritage list in 2001. JNP covers 1,374 km^2, of which 905 km^2 comprises coastal areas and hawksbill turtles nest at the westernmost beaches, Bahía de las Águilas (4.4 km long) and La Cueva (2.5 km). The park receives c.a. 20,000 visitors per year (Wielgus et al. 2010). DENP is the second largest coastal national park in the DR, and comprises some of the southeastern DR and Saona Island at approximately 110 km^2. Hawksbill nesting activity is concentrated on 12 sandy beaches (altogether comprising a total of 15 km) interspersed with rocky areas (Revuelta et al. 2013). The adjacent neritic zone is characterized by a wide continental shelf comprising fringing reefs and rocky bottom communities mixed with seagrass beds. Despite DENP's designation as a national park, the government of the DR allows small-scale artisanal fisheries to occur within its boundaries, mainly for local consumption. However, overfishing has affected the area with populations of invertebrates (conch and spiny lobster) and reef fish highly depleted (Chiappone et al. 2000). In addition, Saona Island is one of the most popular tourist destinations in the DR with up to one thousand visitors a day (MacLeod 2001) resulting in a heavy traffic of motor boats, especially at the west part of the island.

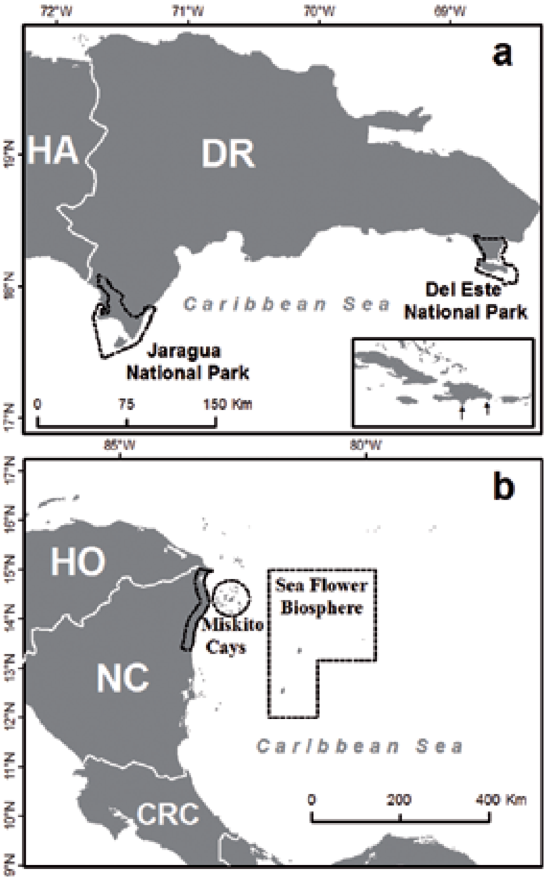

Figure 8.1. Study area. a) Map of DR indicating hawksbill nesting and foraging areas at the protected areas of JNP (southwest DR) and DENP (southeast DR, including Saona Island) where turtles were tagged. B) Hawksbill foraging areas at the Western Caribbean. Boundary lines for MPAs in the Caribbean Sea are shown. HA: Haiti; HO: Honduras; NC: Nicaragua; CRC: Costa Rica.

Marine turtle tracking and satellite data filtering

We satellite-tagged a total of nine nesting hawksbill turtles, eight on Saona Island in August-September 2008 and September-December 2009; and one (Ei8) in JNP in September 2009. Turtles ranged in size from 81.0 to 94.0 cm CCL (Mean ± SD: 87.3 ± 4.4 cm; Table 8.1). No turtles had previous tags. Tags were Wildlife Computers SPOT5 tags (n = 5) and Sirtrack Kiwisat 101 tags (n = 4). To attach the units, we detained each turtle inside a portable wooden corral following nesting or nesting attempts, intercepting them in their way back to the sea. The carapace of each turtle was prepared by scrubbing to remove epibionts, sand, and cleaning with acetone. The units were attached to the second vertebral scute of the turtles' carapace with 2-part epoxy and covered with a layer of anti-fouling paint (Blumenthal et al. 2006). Before attaching the transmitter, each turtle was measured (curved carapace length) and tagged with small metal Inconel tags to the front flippers. Once tagging was complete we removed the corral, allowing the turtle to return to the sea.

For all transmitters, data were downloaded from the ARGOS satellite system and subsequently analyzed with the satellite tracking and analysis tool (STAT, Coyne and Godley 2005) to archive and filter location data. For each reported location, ARGOS calculates a measure of accuracy using six "location classes" (LC): 3, 2, 1, 0, A, and B). As in previous hawksbill telemetry studies (Cuevas et al. 2008; Gaos et al. 2012) the majority of our LCs were categorized as B, thus we considered the positions LC: 3, 2, 1, A and B to avoid loss of relevant location data. These LCs were retained and filtered to remove biologically unrealistic speeds (>5 km h^{-1}; see Luschi et al. 1998), turning angles (<25° were removed) and elevations (>0 metres above sea level).

Turtle's core-use areas and distribution within MPAs

To minimize autocorrelation in spatial analysis we generated mean daily locations for each turtle from the filtered locations (Hart et al. 2010, 2013). To determine core-use areas, foraging and nesting data were extracted using displacement plots and assessed separately (Hawkes et al. 2012). Home range size was estimated using minimum convex polygons (MCP), a non-statistical measure which encapsulates the area used by an individual within a polygon formed by joining the outer-most sighting positions (Burt 1943). MCP is a simple calculation that allows for comparisons between studies (Hooge et al. 1999), but is unable to describe fine-scale movements and preferred area used within the polygon. It may also be inflated by inaccurate yet plausible data outside the true area of utilisation (Laver and Kelly 2008). Therefore, core-use areas were identified using fixed kernel density estimation (KDE) with individual kernel contours delineated using a smoothing factor (h) of 1.5 for internesting and 2.5 for foraging. Density distributions were represented on the maps by the 50% and 90% utilization distribution (UD) contours. We used a 90% KDE to estimate the overall home foraging and internesting range of a turtle and a 50% KDE to represent the core area of activity (Powell 2000). For turtles tracked through two subsequent nesting seasons KDE was calculated separately for each internesting period. Following the methodology of Hart et al. (2013) we did not estimate KDEs for turtles that transmitted for less than 20 days (i.e., had less than 20 mean daily locations).

Common use areas

Common core-use areas were generated for all turtles combined (i.e. where multiple turtles spent time during the internesting and foraging periods). These areas were determined using individual 90% and 50% kernel-density estimates (KDEs).

To analyse the location of turtles with respect to MPAs, filtered turtle location data were overlaid on the World Database on Protected Areas (www.wdpa.org). Site fidelity was quantified using a residency index (Mason and Lowe 2010) calculated by dividing the number of days a female was detected within MPAs' boundaries by the number of days the female was monitored (i.e. internesting period). Values range from 0, indicating no residency, to 1 indicating a high degree of residency. All spatial analyses were carried out in ArcGIS 10 (ESRI 2010).

Characterization of turtle's movements during internesting period

In the present study, and similar to methodology used by Rees et al. (2010), Tucker (2010) and Maxwell et al. (2011), nesting activity was extracted from the tracking data by evaluating the following criteria, classing a nesting event as having occurred when: (a) locations were within 1km of the coastline; (b) the turtle made directed on shore movement; (c) movements occurred within the known internesting intervals for hawksbill turtles (~15 days, Beggs et al. 2007); (d) high-Argos LCs 3, 2, and 1 within a short time span.

RESULTS

Overall transmission success rate (mean number of locations received per day, referred to as 'MDL' from here on) ranged between 1.8 and 5.6 locations per day (SD range 1.3 to 2.7 locations per day) during the internesting period, and 1.0 to 4.2 (SD range 1.2 to 2.0 locations per day in foraging grounds; Table 8.2). Transmitter duration was variable: Turtle Ei1 was only tracked during the internesting period; turtles Ei4 and Ei8 departed immediately following device attachment and were therefore only tracked during the foraging period; the six other turtles (Ei2, Ei3, Ei5, Ei6, Ei7 and Ei9) were tracked during both internesting and foraging (Table 8.1). Transmitters attached to three turtles (Ei2, Ei3 and Ei9) functioned for particularly long periods, permitting insight into multi-year space use. These three turtles were tracked from arrival at their foraging grounds until their departure to breed and nest again in the DR.

Insight into internesting period behaviour

For the internesting period, we analysed a total of 370 tracking days from the seven turtles for which we had internesting data (see above). Individual tracking durations during the internesting period ranged from 0 to 64 days (Mean ± SD: 37.4 ± 22.7 days, Table 8.1). Excluding turtles Ei4 and Ei8 as well as turtle Ei9 in its first nesting season that immediately left coastal waters after laid the last clutch, we estimated that hawksbill females migrate from the internesting area after laying their last clutch of the season a mean (±SD) of 4.3 ± 5.8 days later (range: 0-14 days; Table 8.1). Twenty nesting events were inferred from seven turtles (including two turtles that were recorded nesting in two different years) (Table 8.1). We estimated a mean (±SD) time between nesting events (internesting interval) of 15.5 ± 0.9 days (range: 14-17 days). Turtles E1, Ei2 and Ei3 did not successfully nest at the time of transmitter deployment, thus internesting interval was determined from the first inferred nesting date instead. Mean (±SD) minimum clutch frequency of 7 females for which we have records was 2.9 ± 0.6 clutches (range: 2 to 4 clutches; Table 8.1).

During the internesting period all turtles remained in the territorial waters of Dominican Republic, mostly over the continental shelf (<200 m depth; Figure 8.2). Turtles spent most of their time in areas characterized by relatively shallow waters (KDE 90% for all turtles over water less than 100 m deep). Turtles Ei2 and Ei5 made excursions beyond the shelf (e.g. to the 1000 m isobath, Figure 8.2b and 8.2d). Turtles were usually located 1.4 to 4.3 km from the coast (SD range: 2.1-5.7 km) and mean maximum distance from the coast was 22.4 km (SD 9.2; range: 13 to 39 km; Table 8.1). Analyses of the turtles daily locations show that during the internesting period, of the 370 total tracking days, 306 (82.7%) were within the DENP's borders. Excluding turtle Ei5, turtle's residency index ranged from 0.65 to 1.0 (mean ± SD: 0.84 ± 0.1) indicating a high degree of use of the protected area.

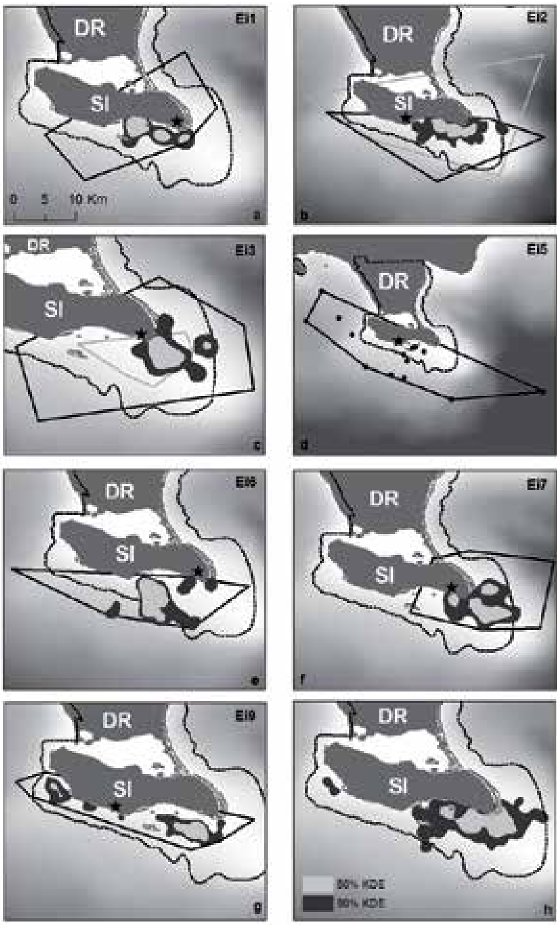

Figure 8.2. *Eretmochelys imbricata.* Individual internesting areas in DENP indicated by minimum convex polygons (MCP) and 90% (black) and 50% (light grey) utilization distributions (UDs) for (a) turtle Ei1; (b) turtle Ei2; (c) turtle Ei3; (d) turtle Ei5; (e) turtle Ei6; (f) turtle Ei7 (g) turtle Ei9; (h) Common core-use area of seven turtles tracked during the internesting period identified using 50% and 90% utilization distributions (UDs). For turtles Ei2 and Ei3 two MCP areas: 1st internesting period (black) and 2nd internesting period (light grey) are shown. Ei2 and Ei3 UDs depict overlap zones 50% and 90% KDEs of two internesting periods. DR: Dominican Republic. SI: Saona Island. Coral reef ecosystems are indicated by hatched areas and marine protected areas by dashed black lines. Stars indicate tagging location. Dashed light grey lines: 200 m and 1000 m bathymetric contour.

Table 8.1. Summary information of Saona nesting hawksbill behaviour and habitat utilisation during internesting period. CCL: curved carapace length. IN: internesting. IN tracking duration: defined as time from tagging date until the last internesting location. Inferred nesting dates: nesting emergences inferred from satellite-tracking data during the IN interval. IN interval: number of days between two consecutive nesting events. Turtles Ei3 and Ei9 stopped sending signals during their second internesting period in 2011 thus we could not estimate time in the IN after the last nest. Turtles Ei4, Ei8 immediately left coastal waters; therefore no internesting interval and clutch frequency could be estimated for these turtles.* Turtles tracked during two successive nesting seasons.

Turtle	Tag date	CCL (cm)	IN tracking duration (days)	Inferred nesting dates	IN interval (days) Mean ± SD	Clutch frequency	Days in IN after last nest	Swim speed (km h^{-1}) Mean ± SD	Max distance offshore (km)
Ei1	11/08/2008	84	34	15/08/2008 30/08/2008 14/09/2008	15.0 ± 0.0	3	1	1.4 ± 1.2	17
Ei2*	28/08/2008	90	60	16/09/2008 29/09/2008 14/10/2008	14.0 ± 1.4	3	14	1.5 ± 1.3	28
Ei2	–	–	–	–	–	–	–	–	33
Ei3*	29/08/2008	90	48	15/09/2008 30/09/2008 14/10/2008	15.3 ± 1.4	3	1	1.2 ± 1.3	25
Ei3	–		52	24/07/2011 10/08/2011	17.0	2	–	0.7 ± 0.8	13
Ei4	19/09/2008	94	0	–	–	–	0	–	–
Ei5	30/09/2008	92	28	15/10/2008	15.0	2	13	2.4 ±2.7	39
Ei6	27/10/2008	81	51	28/11/2008 14/12/2008	16.0	3	2	1.4 ±1.6	17
Ei7	01/11/2008	84	37	18/11/2008 03/12/2008	16.0 ± 1.4	3	4	1.4 ± 1.5	16
Ei8	07/08/2009	84	0	–	–	–	0	–	–
Ei9	01/09/2009	87	64	02/07/2011 18/07/2011 02/08/2011 18/08/2011	15.7 ± 0.6	4	–	0.8 ± 1.0	14

Core use internesting areas

We calculated MCPs for these 7 turtles, two of which transmitted data for a second nesting season, resulting in 9 MCPs (turtles Ei2 and Ei3; Table 8.2 and Figure 8.2). Internesting areas occupied by the turtles ranged from 51.5 to 644.3 km^2 (mean ± SD: 254.5 ± 173.5, Table 8.2). The 90% KDEs (n = 8) for the 6 turtles for which internesting core areas were calculated were much smaller and ranged from 19.6 to 86.0 km^2 (mean ± SD: 49.1 ± 20.1 km^2, Table 8.2); and the mean ± SD 50% KDEs area was 13.7 ± 4.9 km^2 (5.4 to 21.1 km^2, Table 8.2). With the exception of one turtle (Ei2b) during internesting there was no significant variation in home range size among the 7 hawksbill turtles tracked during their internesting period (Figure 8.3), with large portions of MCPs and KDE overlapping (Figure 8.2). Turtles Ei2 and Ei3 showed internesting habitats with similar size and location during their two nesting tracked seasons (Figure 8.2b and 8.2c). There were no correlations between turtle size and the number of days the turtles spent in internesting areas (Pearson's t = -0.7, df = 5, p= 0.5) and between turtle size and the internesting area size (Pearson's t = -1.2, df = 5, p= 0.3).

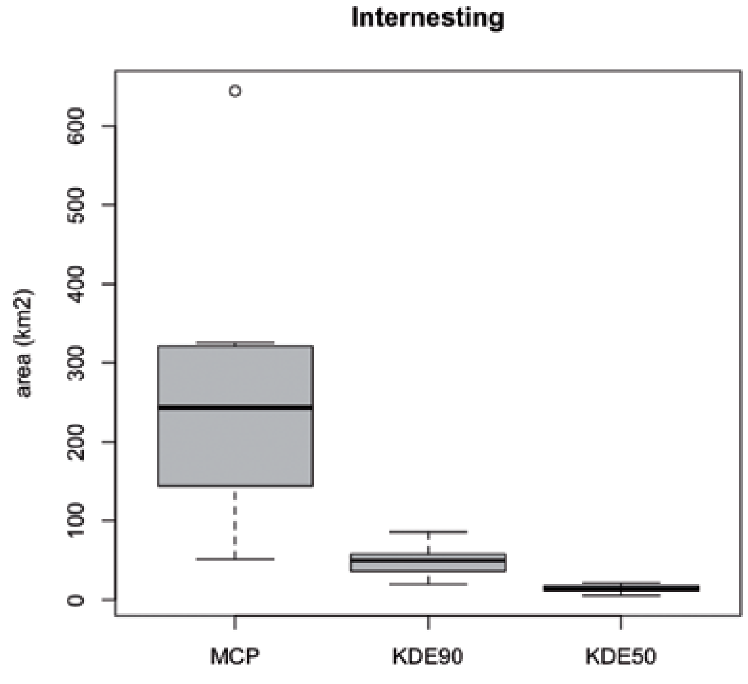

Figure 8.3. Home range size among the 7 hawksbill turtles tracked during their internesting period in the Dominican Republic. MCP: Minimum Convex Polygons; KDE: Kernel Density Estimation (see Material and Methods section for descriptions).

Table 8.2. Transmission success (mean number of locations received per day, mdl), Minimum convex polygon (MCP) and Kernel density estimation (KDE) for hawksbill turtles in their internesting and foraging grounds. BH: Bahamas; CO: Colombia; DR: Dominican Republic; HO: Honduras; NC: Nicaragua. Turtle Ei1 stopped transmitting before arrive the foraging ground. Turtle Ei2 was tracked during two nesting and foraging seasons. Ei3 was tracked during two nesting seasons and one foraging season. Turtles Ei4 and Ei8 departed immediately following device attachment. Turtle Ei5 had less than 20 mean daily locations (n= 16) during the internesting period for KDE analysis. Minimum Convex Polygon sizes in foraging grounds also reported in Hawkes et al. 2012.

Internesting period						Foraging period			
Turtle	mdl	MCP (Km^2)	90% KDE (Km^2)	50% KDE (Km^2)	mdl	MCP (Km^2)	90% KDE (Km^2)	50% KDE (Km^2)	Maritime boundaries
Ei1	3.1±2.0	266.6	32.5	12.2	–	–	–	–	–
Ei2a	1.9±1.8	321.1	57.9	15.1	2.2±1.4	5007	173.1	21.4	NC/HO
Ei2b	–	644.3	55.6	14.2	–	38	14.2	3.9	NC
Ei3a	3.8±2.6	325.4	39.9	11.8	2.5±1.5	25561	257.1	24.9	NC/COL
Ei3b	–	51.5	19.6	5.4	–	–	–	–	–
Ei4	–	–	–	–	1.8±1.2	9330	201.2	45.3	NC/HO
Ei5	2.1±1.8	109.5	–	–	1.0±1.8	314	24.9	13.2	DR
Ei6	2.3±1.3	184.7	57.9	19.2	2.0±1.4	1704	63.6	6.9	DR
Ei7	1.8±1.3	243.0	86.0	21.1	1.3±1.2	3618.8	57.2	21.1	BH
Ei8	–	–	–	–	2.4±1.4	252.1	46.8	4.1	HO
Ei9	5.6±2.7	144.1	43.8	10.4	4.2±2.0	15522	297.8	34.5	NC/HO

Core use foraging areas and MPAs

We also calculated MCPs and 90% and 50% KDEs for turtles in their foraging grounds (Table 8.2). Foraging areas of five females (Ei2, Ei3, Ei4, Ei8, Ei9) were located along the waters off Honduras and Nicaragua (Figure 8.3a). In those waters there are two main protected areas, the Miskito Cays and the Seaflower MPA (part of the Seaflower Biosphere Reserve, belonging to Colombia). The Miskito Cays (27 km^2) is an archipelago located off shore in the North-Eastern Caribbean coast of Nicaragua (Figure 8.1). The seagrass beds and reefs in the cays are among the Atlantics's greatest foraging grounds for green (*Chelonia mydas*) and hawksbill marine turtles (Bjorndal and Bolten 2003). The Seaflower MPA, located in the south-western Caribbean sea (Figure 8.1) is the largest MPA in the wider Caribbean (65,000 km^2) protecting mangroves, seagrass beds and the largest and most productive coral reefs in the region (Taylor et al. 2013).With the exception of turtles Ei3 and Ei4 with residency index of 0.02 and 0.91 respectively (Seaflower Biosphere Reserve and Miskitos Cays respectively), the rest of the turtles were not located in protected waters for any of their tracked foraging period (Figure 8.4a to 8.4e). After nesting in Saona, turtles Ei5 and Ei6 remained in waters of the DR within coastal reef ecosystems (Figure 8.5a and 8.5b). Turtle Ei5 stayed inside the JNP (residency index: 0.91; Figure 8.5a). Turtle Ei6 foraging area was located in waters adjacent to Bahía de las Calderas (southern DR coast) outside of marine protected areas (Figure 8.5b). Turtle Ei7 travelled to the northwestward to the Bahamas where its core use area was not in protected waters (Figure 8.5c).

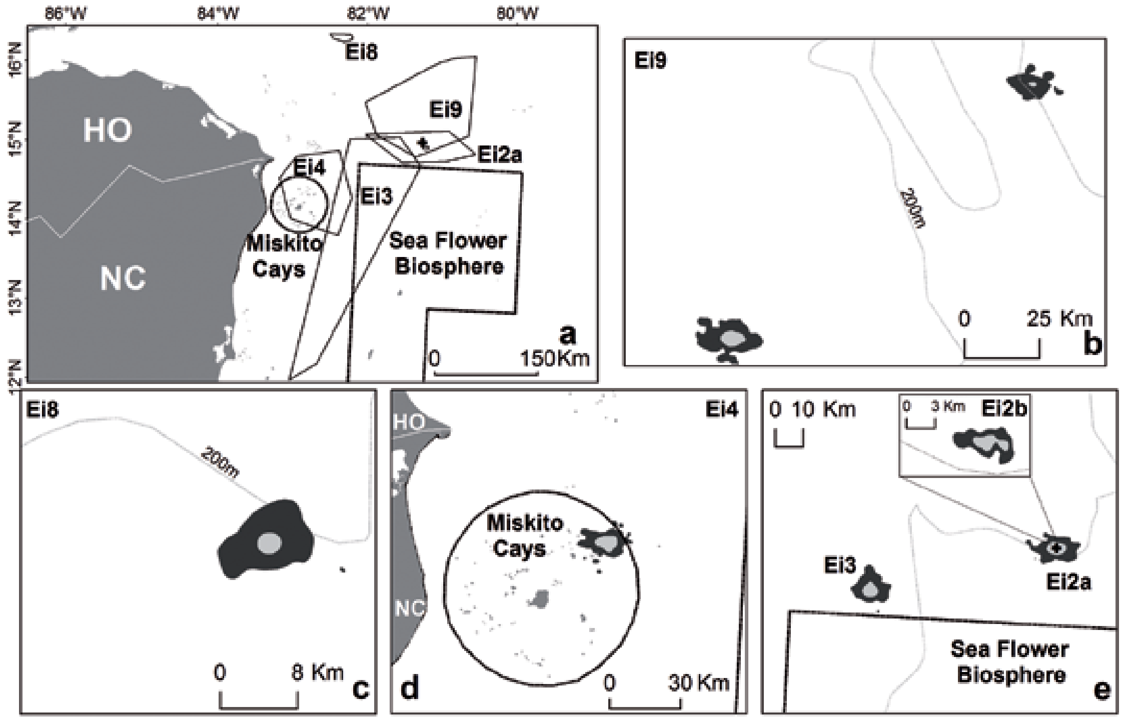

Figure 8.4. a) MCPs of hawksbill turtles at Nicaragua and Honduras foraging area. (+): MCP of Ei2b (2012 foraging period). Core-use areas defined by 90% (black) and 50% (light grey) utilization distributions (UDs) plotted with adjacent MPAs of Miskito Cays and Seaflower (dashed black lines) for b) turtle Ei9 which occupied two separated areas during the same foraging period; c) turtle Ei8 d) turtle Ei4 and e) turtles Ei2a (2008 foraging period) and Ei3 (inset: UDs for turtle Ei2b during 2012 foraging period, which overlapped with UDs in 2008). Note different scales. Dashed light grey line: 200 m bathymetric contour.

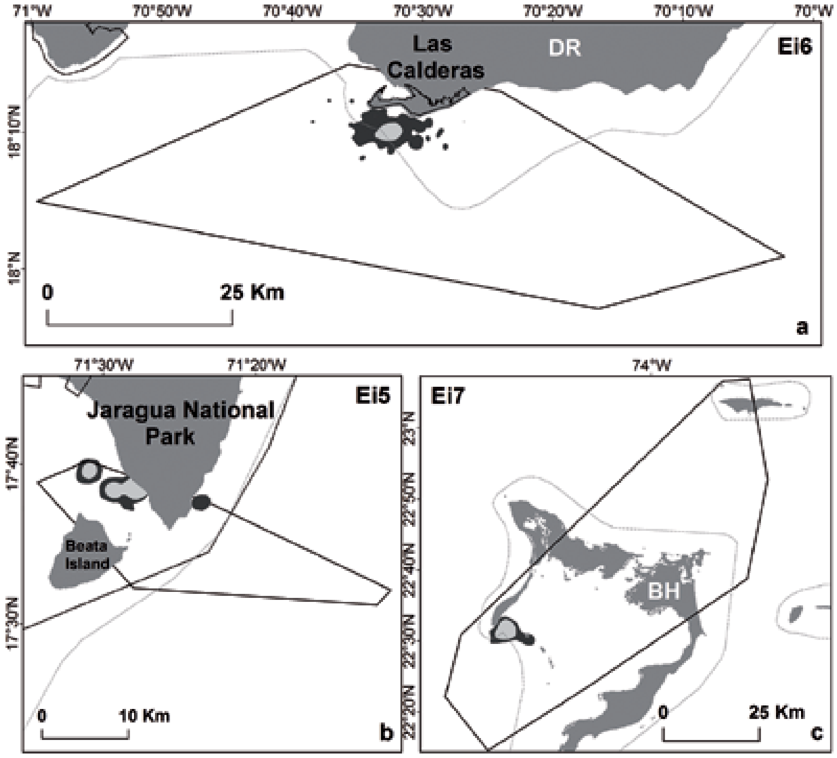

Figure 8.5. Foraging core use areas defined by MCPs and 90% (black) and 50% (light grey) utilization distributions (UDs) of a) turtle Ei6 near shore of Las Calderas Natural Monument (south DR) b) turtle Ei5 in waters inside of JNP boundaries (Southwest DR); and c) turtle Ei7 in waters of Bahamas. Note different scales.

Common-use areas

The internesting common-use area 90% and 50% KDE was 81.7 km^2 and 32.2 km^2 respectively (Figure 8.2h). This common-use area was situated on coral reefs at the eastern-most tip of the island and the individual home ranges for all seven turtles overlapped, showing commonality across turtles and years. Overall, the common core-use area during nesting was situated inside the DENP's boundaries (Figure 8.2h). There was, however, no overlap in foraging grounds used by females tracked in the same year; but, there was overlap in resident areas utilised by one turtle that was tracked in two different foraging period (Ei2) used same location in both years (Figure 8.4e).

DISCUSSION

Core use areas and MPAs

Home range estimations provide knowledge of marine turtles' core areas of activity, underscoring hotspots for their protection (Scott et al. 2012; Schofield et al. 2013). The present study has permitted insight into the effectiveness of some existing MPAs in the protection of the nesting hawksbills. It should be noted that the number of tracked turtles studied in the present paper represent ca. 40% of the total annual nesting stock in the Dominican Republic (Revuelta et al. 2012). Overall, during internesting intervals turtles were located within MPAs while during the foraging period they were mostly (78.0 % of total foraging days tracked) outside of any MPA.

Our results reveal that during their internesting period hawksbill turtles in Saona Island are mostly within the maritime limits of the DENP, highlighting the importance of effective management of this area for their conservation. Although the spatial extent of the DENP offers a good opportunity to enhance the protection for this critical breeding ground, turtles face multiple anthropogenic threats. Firstly, the expansion of tourism industry has increased the boat traffic around Saona Island, particularly in the west part, and also increasing the pressure on coral reefs due to direct pollution (Wilkinson 2000). Secondly, artisanal fisheries have essentially uncontrolled access to the park, resulting in the depletion of large reef fish, conch and lobster populations (Chiappone et al. 2000). In addition, illegal capture of adult turtles by illegal fisherman has also been documented (Revuelta et al. 2013).

Hawksbill turtles tagged in Saona show a range of migratory strategies, with some turtles remaining near nesting sites in the DR waters and others migrating to international foraging grounds (Hawkes et al. 2012). Most of the turtles that migrated internationally foraged in waters off Nicaragua and Honduras (n = 5), which has already been confirmed as a preferred foraging ground for hawksbills nesting in other Caribbean areas such as Costa Rica, Eastern Caribbean and Cuba (Troëng et al. 2005; Horrocks et al. 2011; Moncada et al. 2012). Of these 5 turtles just one (Ei4) spent a large proportion of the time inside a protected area (Miskito Cays). However, the effectiveness of the protection in this area is questionable as a result of legal and illegal marine turtle fisheries (Bräutigam and Eckert 2006). The lack of protection in the waters off Nicaragua and Honduras thus poses a potentially significant conservation problem for many Caribbean hawksbill nesting populations. In the Bahamas, there are eight marine national parks, and direct harvest of marine turtles is probably considerably less common. However, protected waters make up a minority of the 630,000 km^2 exclusive economic zone of the Bahamas (less than 1%) and thus it is not surprising that the turtle foraging here did so outside of protected areas.

Turtle Ei5 stayed in DR waters, spending most of the time (91 % of the total days tracked) within JNP maritime limits, which supports previous studies about the importance of this area for marine turtle conservation in the country, not only as a foraging ground for juveniles and adults but also as a nesting area (León and Diez 1999; Revuelta et al. 2012). The second potential foraging ground in DR was located in the adjacent waters of the Natural Monument Bahía de Calderas in the south coast of the country. This protected area encompasses around 15 km of sand dunes but does not yet incorporate the sea. Sporadic nesting by *Chelonia mydas* has also been detected in these beaches (Y.M. León pers. obs.). Despite the evident importance of this area for biodiversity conservation, and its protected status, it is threatened by the extraction of sand for commercial purposes and illegal construction by the hotel industry, as well as indiscriminate fishing activities (Perdomo et al. 2010). Such lack of enforcement of conservation actions in protected areas in the country prevents an effective protection of these critical habitats for marine turtles and is an urgent target for improved conservation.

Internesting behaviour

Nesting hawksbills of Saona Island remained in the adjacent waters to their nesting beaches using small home range areas during internesting intervals. Core activity areas occurred in shallow waters mainly within 200 m isobaths and associated with coral reefs. Our results support previously described hawksbill internesting behaviour observed in other internesting areas throughout the Caribbean (van Dam et al. 2008; Marcovaldi et al. 2012; Walcott et al. 2012). Regardless of the nesting location on the beach, core-use areas were situated on coral reefs at the eastern-most tip of the island. Turtle preference for particular internesting areas has been related to adequate resources and quality of the habitat occupied (Hart et al. 2010). The abundance of fringing reef systems inside DENP's marine area likely accounts for hawksbill turtle affinity for this area. The abundance of sheltered resting sites could be an important resource, helping nesting females to preserve energy during their reproductive period (Houghton et al. 2008; Hart et al. 2010). Although it may not the case everywhere (e.g., Zbinden et al. 2007), we hypothesize that the huge number of tourist boats in the vicinity of nesting beaches in the west part of Saona, could lead to avoidance behaviour.

Internesting activity patterns for Saona hawksbill turtles were similar to results for hawksbill turtles satellite tracked during nesting periods in other locations. However it should be noted that since tagging date was the only confirmed nesting event, nesting dates were inferred by turtle tracks (see Materials and Methods section), thus nesting dates and internesting intervals are approximated. Internesting interval ranged between 14 to 17 days described for the species in closer areas (Beggs et al. 2007; van Dam et al. 2008). Tracking-derived minimum clutch frequency, which is unable to account for nests laid prior to transmitter attachment, ranged from 2 to 4 clutches, which is in accordance with conspecifics elsewhere (Witzell 1983). Previous researchers have hypothesized that hawksbill turtles may not forage during internesting intervals and at the end of the nesting season, females would need to migrate back to foraging grounds to restore their energy reserves (van Dam et al. 2008). The present study seems support this hypothesis since turtles remained in the internesting area only for a few days following their final nesting event. It is possible that the longer post-nesting periods inferred for turtles Ei2 and Ei5 may have been due to repeat nesting that we were unable to identify, or because some foraging may take place.

Conclusions and conservation recommendations

In this study, we describe DR hawksbill nesting turtles use of MPAs at nesting and feeding grounds, adding information to the use of MPAs by this species in the Caribbean region. This information highlights the significance of protected areas in the DR for internesting and foraging hawksbills, showing the need to enforce existing legislation of the protected areas in the country. The turtles use waters inside the protected areas of DENP and JNP year-round, areas that are severely threatened by human activities. Hence, efforts must be increased to mitigate illegal fishing in the waters of these parks. In the case of waters around Saona we recommend the creation of near shore zone of maximum protection that would enhance the protection of the rookery in this highly used area (i.e., by restricting the boat traffic in this zone). Likewise, we propose the expansion of Las Calderas Natural Monument boundaries offshore due to its importance not only for turtles but also for other species and ecosystems, as well as enhanced enforcement of existing regulations in the Monument. The present study also corroborates that the waters off Nicaragua and Honduras are exceptionally important foraging areas for hawksbills in the Caribbean. We recommend that MPAs for marine turtle conservation in the region should be reassessed as a priority.

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REFERENCES

Beggs JA, Horrocks JA and Krueger BH (2007) Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies. Endangered Species Research, 3, 159 - 168.

Bjorndal KA and Bolten AB (2003) From ghosts to key species: restoring sea turtle populations to fulfill their ecological roles. Marine Turtle Newsletter, 100, 16 - 21.

Bjorndal KA and Jackson JBC (2003) Role of sea turtles in marine ecosystems: reconstructing the past. In: Lutz PL, Musick JA and Wyneken J (eds). Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, Florida, pp 259 - 273.

Blanco GS, Morreale SJ, Bailey H, Seminoff JA, Paladino FV and Spotila JR (2012) Post-nesting movements and feeding grounds of a resident East Pacific green turtle *Chelonia mydas* population from Costa Rica. Endangered Species Research, 18, 233 - 245.

Blumenthal JM, Solomon JL, Bell CD, Austin TJ, Ebanks-Petrie G, Coyne MS, Broderick AC and Godley BG (2006) Satellite tracking highlights the need for international cooperation in marine turtle management. Endangered Species Research, 2, 51 - 61.

Bograd SJ, Block BA, Costa DP and Godley BJ (2010) Biologging technologies: new tools for conservation. Introduction. Endangered Species Research, 10, 1 - 7.

Bräutigam A and Eckert KL (2006) Turning the tide: exploitation, trade and management of marine turtles in the Lesser Antilles, Central America, Colombia and Venezuela. TRAFFIC International, Cambridge.

Burt WH (1943) Territoriality and home range concepts as applied to mammals. Journal of Mammalogy, 24, 346 - 352.

Chiappone M, Sluka R and Sullivan Sealey K (2000) Groupers (Pisces: Serranidae) in fished and protected areas of the Florida Keys, Bahamas and northern Caribbean. Marine Ecology Progress Series, 198, 261 - 272.

Costa DP, Breed GA and Robinson PW (2012) New insights into pelagic migrations: implications for ecology and conservation. Annual Review of Ecology, Evolution and Systematics, 43, 73 - 96.

Coyne MS and Godley BJ (2005) Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series, 301, 1 - 7.

Cuevas E, Abreu-Grobois FA, Guzman-Hernandez V, Liceaga-Correa MA and van Dam RP (2008) Post-nesting migratory movements of hawksbill turtles *Eretmochelys imbricata* in waters adjacent to the Yucatan Peninsula, Mexico. Endangered Species Research, 10, 123 - 133.

Edgar GJ (2011) Does the global network of marine protected areas provide an adequate safety net for marine biodiversity? Aquatic Conservation: Marine and Freshwater Ecosystems, 21, 313 - 316.

ESRI (2010) ArcGIS 10.0 GIS. Redlands, CA.

Gaos AR, Lewison RL, Wallace BP, Yanez IL, Liles MJ, Nichols WJ, Baquero A, Hasbun CR, Vasquez M, Urtega J and Seminoff JA (2012) Spatial ecology of critically endangered hawksbill turtles *Eretmochelys imbricata* implications for management and conservation. Marine Ecology Progress Series, 450, 184 - 194.

Godley BJ, Blumenthal JM, Broderick AC, Coyne MS, Godfrey MH, Hawkes LA and Witt MJ (2008) Satellite tracking of sea turtles: Where have we been and where do we go next? Endangered Species Research, 4, 3 - 22.

Halpern BS (2014) Conservation: Making marine protected areas work. Nature, 506, 167 - 168.

Hammerschlag N, Gallagher AJ and Lazarre DM (2011) A review of shark satellite tagging studies. Journal of Experimental Marine Biology and Ecology, 398, 1 - 8.

Hart KM and Hyrenbach KD (2010) Satellite telemetry of marine megavertebrates: The coming of age of an experimental science. Endangered Species Research, 10, 9 - 20.

Hart KM, Zawada DG, Fujisaki I and Lidz BH (2010) Inter-nesting habitat-use patterns of loggerhead sea turtles: enhancing satellite tracking with benthic mapping. Aquatic Biology, 11, 77 - 90.

Hart KM, Lamont MM, Sartain AR, Fujisaki I and Stephens BS (2013) Movements and habitat-use of loggerhead sea turtles in the Northern Gulf of Mexico during the reproductive period. PLoS ONE. doi:10.1371/journal.pone.0066921.

Hawkes LA, Tomás J, Revuelta O, León YM, Blumenthal JM, Broderick AC, Fish M, Raga JA, Witt MJ and Godley BJ (2012) Migratory patterns in hawksbill turtles described by satellite tracking. Marine Ecology Progress Series, 461, 223 - 232.

Hoffman M, Belant JL, Chanson JS, Cox NA, Lamoreux J, Rodrigues ASL, Schipper J and Stuart SN (2011) The changing fates of the world's mammals. Philosophical Transactions of the Royal Society B, 366, 2598 - 2610.

Hooge PN, Eichenlaub WM and Solomon EK (1999) The animal movement program. USGS, Alaska Biological Science Center. http://www.absc.usgs. gov/glba/gistools.

Horrocks JA, Krueger BH, Fastigi M, Pemberton EL and Eckert KL (2011) International movements of adult female hawksbill turtles (*Eretmochelys imbricata*): first results from the Caribbean's marine turtle tagging centre. Chelonian Conservation and Biology, 10, 18 - 25.

Houghton JDR, Cedras A, Myers AE, Liebsch N, Metcalfe JD, Mortimer JA and Hays GC (2008) Measuring the state of consciousness in a free-living diving sea turtle. Journal of Experimental Marine Biology and Ecology, 356, 115 - 120.

IUCN (2013) The IUCN red list of threatened species. http://www.iucnredlist.org. Accessed 11 November 2013.

Laver PN and Kelly MJ (2008) A Critical Review of Home Range Studies. Journal of Wildlife Management, 72, 290 - 298.

León YM and Diez CE (1999) Population structure of hawksbill turtles on a foraging ground in the Dominican Republic. Chelonian Conservation and Biology, 3, 230 - 236.

León YM and Bjorndal KA (2002) Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. Marine Ecology Progress Series, 245, 249 - 258.

Luschi P, Hays GC, Del Seppia C, Marsh R and Papi F (1998) The navigational feats of green sea turtles migrating from Ascension Island investigated by satellite telemetry. Proceedings of the Royal Society B: Biological Sciences, 265, 2279 - 2284.

Macleod DVL (2001) Parks or people? National parks and the case of Del Este, Dominican Republic. Progress in Development Studies, 1, 221 - 235.

Marcovaldi MA, Lopez GG, Soares LS and Mendilaharsu M (2012) Satellite tracking of hawksbill turtles *Eretmochelys imbricata* nesting in northern Bahia, Brazil: turtle movements and foraging destinations. Endangered Species Research, 17, 123 - 132.

Mason TJ and Lowe CG (2010) Home range, habitat use, and site fidelity of barred sand bass within a southern California marine protected area. Fisheries Research, 106, 93 - 101.

Maxwell SM, Breed GA, Nickel BA, Makanga-Bahouna J, Pemo-Makaya E, Parnell RJ, Formia A, Ngouessono S, Godley BJ, Costa DP, Witt MJ and Coyne MS (2011) Using satellite tracking to optimize protection of long-lived marine species: Olive Ridley sea turtle conservation in Central Africa. PLoS ONE.doi:10.1371/journal.pone.0019905.

Meylan AB (1988) Spongivory in hawksbill turtles: a diet of glass. Science, 239, 393 - 395.

Meylan PA, Meylan AB and Gray JA (2011) The ecology and migrations of sea turtles. 8. Tests of the developmental habitat hypothesis. Bulletin of the American Museum of Natural History, 357, 1 - 70.

Moncada FG, Hawkes LA, Fish MR, Godley BJ, Manolis SC, Medina Y, Nodarse G and Webb GJW (2012) Patterns of dispersal of hawksbill turtles from the Cuban shelf inform scale of conservation and management. Biological Conservation, 148, 191 - 199.

Perdomo L, Arias Y, León YM and Wege D (2010) Áreas importantes para la conservación de las aves en la República Dominicana. Grupo Jaragua y el programa IBA-Caribe de BirdLife international. Santo Domingo, República Dominicana.

Powell RA (2000) Animal home ranges and territories and home range estimators. In: Boitani L, Fuller T (eds). Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York, pp 65 - 110.

Read AJ, Drinker P and Northridge S (2006) Bycatch of marine mammals in U.S. and global fisheries. Conservation Biology, 20, 163 - 169.

Rees AF, Saady SA, Broderick AC, Coyne MS, Papathanasopoulou N and Godley BJ (2010) Behavioural polymorphism in one of the world's largest populations of loggerhead sea turtles *Caretta caretta*. Marine Ecology Progress Series, 418, 201 - 212.

Revuelta O, León YM, Feliz P, Godley BJ, Raga JA and Tomás J (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. Oryx, 46, 348 - 358.

Revuelta O, León YM, Aznar FJ, Raga JA and Tomás J (2013) Running against time: conservation of the remaining hawksbill turtle (*Eretmochelys imbricata*) nesting population in the Dominican Republic. Journal of Marine Biological Association UK, 93, 1133 - 1140.

Schofield G, Scott R, Dimadi A, Fossette S, Katselidis KA, Koutsoubas D, Lilley MKS, Pantis JD, Karagouni AD and Hays GC (2013) Evidence-based marine protected area planning for a highly mobile endangered marine vertebrate. Biological Conservation, 161, 101 - 109.

Scott R, Hodgson DJ, Witt MJ, Coyne MS, Adnyana W, Blumenthal JM, Broderick AC, Canbolat AF, Catry P, Ciccione S, Delcroix E, Hitipeuw C, Luschi P, Pet-Soede L, Pendoley K, Richardson PB, Rees AF and Godley BJ (2012) Global analysis of satellite tracking data shows that adult green turtles are significantly aggregated in Marine Protected Areas. Global Ecology and Biogeography, 21, 1053 - 1061.

Taylor E, Baine M, Killmer A and Howard M (2013) Seaflower marine protected area: Governance for sustainable development. Marine Policy, 41, 57 - 64.

Troëng S, Dutton PH and Evans D (2005) Migration of hawksbill turtles *Eretmochelys imbricata* from Tortuguero, Costa Rica. Ecography, 28, 394 - 402.

Tucker AD (2010) Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: implications for stock estimation. Journal of Experimental Marine Biology and Ecology, 383, 48 - 55.

van Dam RP, Diez CE, Balazs GH, Colón Colón LA, McMillan WO and Schroeder B (2008) Sex-specific migration patterns of hawksbill turtles breeding at Mona Island. Endangered Species Research, 4, 85 - 94.

Walcott J, Eckert S and Horrocks JA (2012) Tracking hawksbill sea turtles (*Eretmochelys imbricata*) during inter-nesting intervals around Barbados. Marine Biology, 159, 927 - 938.

Wielgus J, Cooper E, Torres R and Burke L (2010) Coastal Capital: Dominican Republic. Case Studies on the Economic Value of Coastal Ecosystems in the Dominican Republic. Working paper. World Resources Institute, Washington, DC, USA.

Wilkinson CR (2000) Status of Coral Reefs of the World: 2000. Australian Institute of Marine Science, Townsville, Australia.

Witzell WN (1983) Synopsis of biological data on the hawksbill turtle *Eretmochelys imbricata* (Linnaeus, 1766). FAO Fisheries Synopsis No. 137. Pp 78.

Zbinden JA, Aebischer A, Margaritoulis D and Arlettaz R (2007) Insights into the management of sea turtle internesting area through satellite telemetry. Biological Conservation, 137, 157 - 162.

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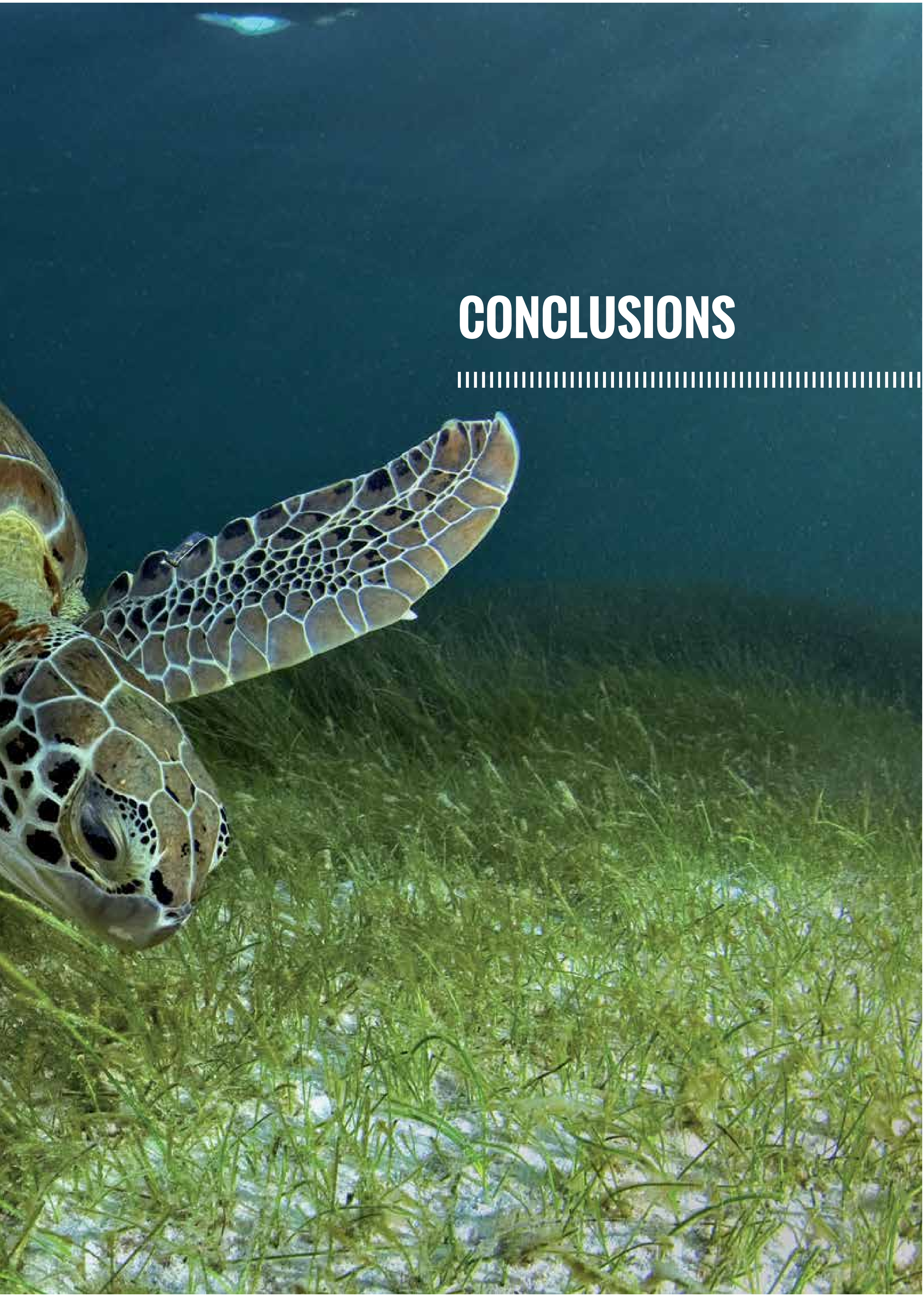

CONCLUSIONS

CONCLUSIONS

The present PhD Thesis analyses the conservation status of marine turtle nesting rookeries in the Dominican Republic. The following section summarizes the main findings and conclusions.

1) This study represents the first detailed assessment of the status of marine turtles nesting in the Dominican Republic, based on surveys during the period 2006-2010, and compares the results with previous studies. Leatherback, hawksbill and green turtles are still nesting in the country; however, unlike previous studies, no loggerhead turtles nesting event was detected during this period.

2) Currently, marine turtle nesting activity is concentrated in the protected areas of Jaragua National Park and Saona Island (Del Este National Park). Nesting events out of these two areas are sporadic. The results of this study suggest that there has been a profound reduction in the abundance of the marine turtle species in the country since the 1980s.

3) The Jaragua National Park, in the southwest of the country, hosts the highest number of clutches per year of leatherback turtles (mean 126.4 ± SD 74.1, range: 17-210), with a total of 632 clutches recorded during the period studied. Nesting season of leatherback turtles extends from March to August, with most emergences occurring in April-June. The estimated annual number of leatherback nesting females varied from 3 to 33 turtles. The size (mean CCL = 147.4 cm) of leatherback turtles in the Dominican Republic is smaller than the global mean size, which could be the sign of a recovering population with a high proportion of neophyte females. Sporadic nesting of hawksbills and green turtles was also recorded in this area.

4) Saona Island, in the southeast of the country, is the main nesting area for the hawksbill turtle (mean 100 ± SD 8.4 clutches per year, range 93-111, n = 400 clutches) and the green turtle (mean 9.2 ± SD 6.2, range 1-15, n = 37). Hawksbill turtles nest all through the year, although most nesting occurs in the period June-November. The estimated annual number of hawksbill nesting females varied from 21 to 25 turtles. The size of nesting hawksbill turtles in the Dominican Republic (mean CCL = 87.2 cm) is similar to the size of the species in other Caribbean regions.

5) The high level of human predation on clutches and coastal development plans are the main threats for the conservation of these marine turtle nesting stocks. To face egg take, an artificial incubation program was carried out.

6) In the Jaragua National Park artificial incubation was carried out in two locations (western and eastern beaches) and hatching success and sex ratio was compared with values of *in situ* incubated clutches. The results showed that in the west, artificial incubation significantly decreased hatching success in clutches and, in the east, the incubation duration increased, which we predict would result in an increase in male production from these clutches.

7) Clutch relocation is currently the only viable conservation option for clutches on eastern beaches due to intense egg take

(~100%), but steps are needed to ensure that natural sex ratio is not distorted. However, on the western beaches, *in situ* clutch incubation seems possible through beach protection.

8) The artificial incubation program conducted in Saona from 2007 to 2010 allowed the release of more than 12000 hawksbill hatchlings. No differences in hatching and emergence success between *in situ* and artificially incubated clutches were found. However, low temperatures and long incubation periods recorded in artificially incubated clutches suggest a bias to male hatchling production. Although artificial incubation is effective in terms of hatchling production, low levels of female production highlights the need to improve this protection measure.

9) We also studied the factors affecting hatching success of leatherback clutches on the western beaches of the Jaragua National Park to inform possible mitigation through clutch translocation in the case of habitat loss. Beach sector, incubation duration, date of lay and clutch size have been found as the main factors driving hatching success.

10) Clutches of leatherback turtles on the western beaches of the Jaragua National Park presented an unusual high hatching success (75.2 ± 21.0%) for the species as compared to other rookeries in the Caribbean (~ 50%). Given the exceptional value of hatching success and the current and potential threats affecting leatherback nesting beaches, additional efforts in regulation and management of this protected area are needed.

11) The sector of La Cueva beach hosts 20% of the total clutches laid on western beaches of the Jaragua National Park and demonstrated the highest hatching success levels. Since this sector is located in the buffer zone out of the Park limits, its inclusion inside the limits of the Jaragua National Park is strongly recommended.

12) Through satellite tracking it was found that during the internesting period hawksbill turtles remained in the territorial waters of Dominican Republic, mostly over the continental shelf (<200 m) in areas characterized by relatively shallow waters close to the corresponding nesting beaches.

13) Overall, the common core-use area of nesting hawksbill turtles of Saona during internesting period was situated inside the Del Este National Park boundaries. Home ranges concentrated in waters at the eastern-most tip of the island, showing similar location and extension both across turtles and years. Efforts should be increased to mitigate illegal fishing and to restrict boat traffic in these waters.

14) During the foraging period, 78.0 % of locations were outside marine protected areas either in waters of the Dominican Republic and in international waters of Bahamas, Nicaragua and Honduras. Our results highlight the significance of different protected areas in the Dominican Republic as hawksbill‘s foraging areas, showing the need of enforcing existing legislation and the expansion of protected areas in the country.

15) The present study also corroborates that the waters off Nicaragua and Honduras are exceptionally important foraging areas for

hawksbill turtles nesting in the Caribbean region, showing the turtles vulnerability in these waters.

16) Despite the importance of protected coastal and marine areas, not only for nesting marine turtles but also for marine biodiversity in the country, the management of these areas is limited due to lack of resources and effort. The recovery of nesting colonies in the country depends on competent management of protected areas, as well as effective implementation of laws prohibiting the consumption and trade of any marine turtle products.


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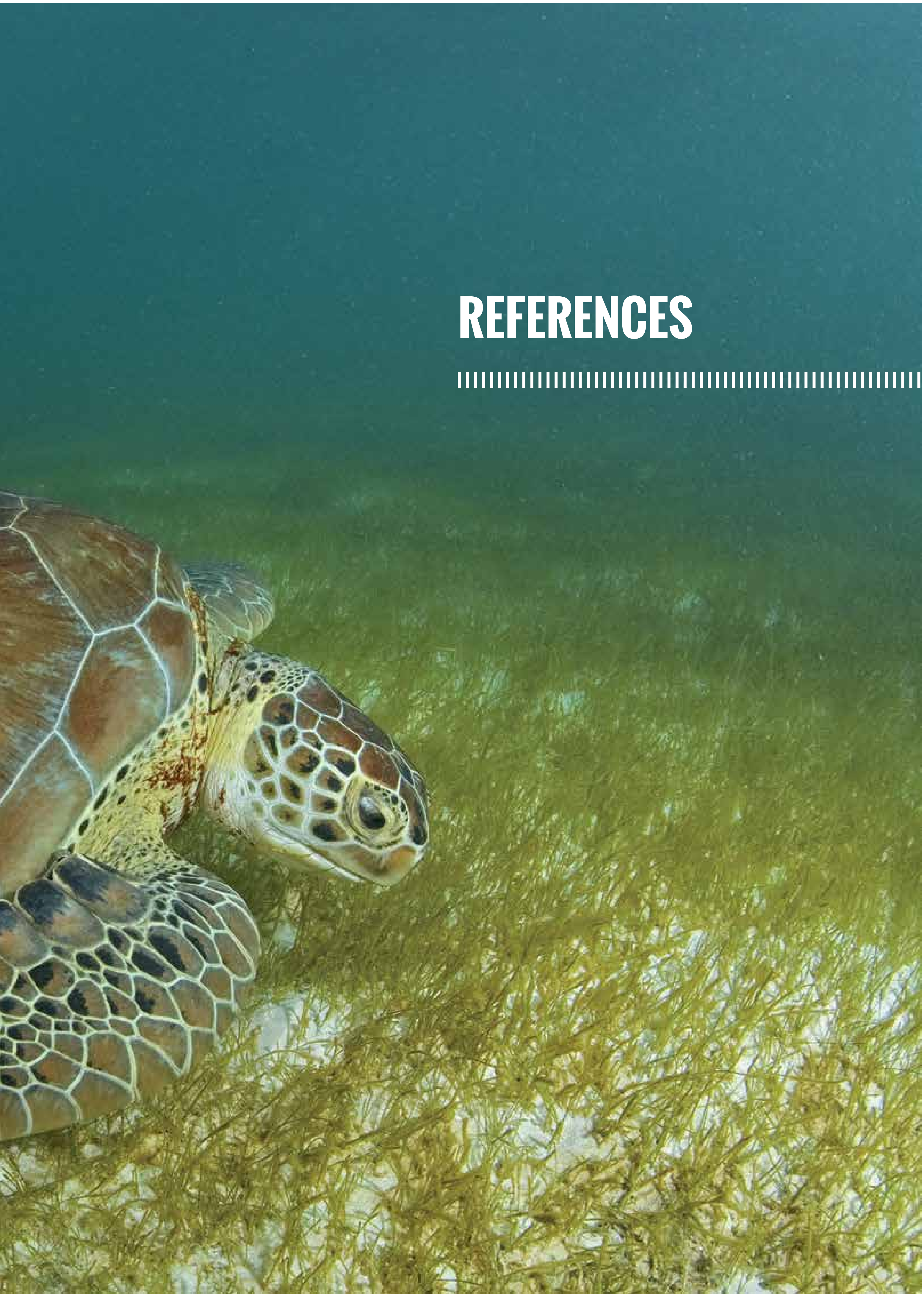
REFERENCES



REFERENCES

Abdulla A, Gomei M, Hyrenbach D, Notarbartolo-di-Sciara G and Agardy T (2008) Challenges facing a network of representative marine protected areas in the Mediterranean: prioritizing the protection of underrepresented habitats. ICES Journal of Marine Science, 66, 22 - 28.

Abella E, Marco A and López-Jurado LF (2007) Success of delayed translocation of loggerhead turtle nests. Journal of Wildlife Management, 71, 2290 - 2296.

Ackerman RA (1980) Physiological and ecological aspects of gas exchange by sea turtle eggs. American Zoologist, 20, 575 - 583.

Ackerman RA (1997) The nest environment and the embryonic development of sea turtles. In: Lutz PL and Musick JA (eds). The biology of sea turtles. CRC Press Inc, Boca Raton FL pp 83 - 106.

Agardy T (2005) Global marine conservation policy versus site level implementation: the mismatch of scale and its implications. Marine Ecology Progress Series, 300, 242 - 248.

Al-Abdulrazzak D and Trombulak SC (2012) Classifying levels of protection in marine protected areas. Marine Policy, 36, 576 - 582.

Allison GW, Gaines SD, Lubchenco J and Possingham HP (2003) Ensuring persistence of marine reserves: catastrophes requiring adopting an insurance factor. Ecological Applications, 13, S8 - S24.

Almeida AP and Mendes SL (2007) An analysis of the role of local fishermen in the conservation of the loggerhead turtle (*Caretta caretta*) in Pontal do Ipiranga, Linhares, ES, Brazil. Biological Conservation, 134, 106 - 112.

Alves RRN and Rosa IML (2007) Biodiversity, traditional medicine and public health: where do they meet? Journal of Ethnobiology and Ethnomedicine. doi:10.1186/1746-4269-3-14.

Anadón-Irizarry V, Wege DC, Upgren A, Young R, Boom B, León YM, Arias Y, Koenig K, Morales AL, Burke W, Perez-Leroux A, Levy C, Koenig S, Gape L and Moore P (2012) Sites for priority biodiversity conservation in the Caribbean Islands Biodiversity Hotspot. Journal of Threatened Taxa, 4, 2806 - 2844.

Andrews HV (2000) Current marine turtle situation in the Andaman and Nicobar islands an urgent need for conservation action. Kachhapa, 3, 19 - 23.

Baillie J and Groombridge B (1996) IUCN Red List of Threatened Animals. IUCN, Gland, Switzerland and Cambridge, UK.

Bass AL, Good DA, Bjorndal KA, Richardson JI, Hillis ZM, Horrocks JA and Bowen BW (1996) Testing models of female reproductive migratory behavior and population structure in the Caribbean hawksbill turtle, *Eretmochelys imbricata*, with mtDNA sequences. Molecular Ecology, 5, 321 - 328.

Bates DM, Maechler M and Dai B (2008) Lme4: Linear mixed-effects models using S4 classes. http://lme4.r-forge.r-project.org/.

Beggs JA, Horrocks JA and Krueger BH (2007) Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies. Endangered Species Research, 3, 159 - 168.

Bell BA, Spotila JR, Paladino FV and Reina RD (2003) Low reproductive success of leatherback turtles, *Dermochelys coriacea*, is due to high embryonic mortality. Biological Conservation, 115, 131 - 138.

Bell CD, Blumenthal JM, Austin TJ, Solomon JL, Ebanks-petrie G, Broderick AC and Godley BJ (2006) Traditional Caymanian fishery may impede local marine turtle population recovery. Endangered Species Research, 2, 63 - 69.

Bell CDL, Solomon JL, Blumenthal JM, Austin TJ, Ebanks-Petrie G, Broderick AC and Godley BJ (2007) Monitoring and conservation of critically reduced marine turtle nesting populations: lessons from the Cayman Islands. Animal Conservation, 10, 39 - 47.

Bjorndal KA (1985) Nutritional ecology of sea turtles. Copeia, 1985, 736 – 751.

Bjorndal KA (1997) Foraging ecology and nutrition of sea turtles. In: Lutz PL and Musick JA (eds). The Biology of Sea Turtles. 15. CRC Press, Boca Raton, FL, pp 397 - 409.

Bjorndal KA (2003) Roles of loggerhead sea turtles in marine ecosystems. In: Bolten AB and Witherington BE (eds). Logger-head sea turtles. Smithsonian Books, Washington, DC, pp 235 - 254.

Bjorndal KA and Bolten AB (2003) From ghosts to key species: restoring sea turtle populations to fulfill ecological roles. Marine Turtle Newsletter, 100, 16 – 21.

Bjorndal KA and Jackson JBC (2003) Roles of sea turtles in marine ecosystems: reconstructing the past. In: Lutz PL, Musick JA and Wyneken J (eds). The Biology of Sea Turtles Volume II. CRC Press, Boca Raton, pp 259 – 273.

Bjorndal KA, Carr A, Meylan AB and Mortimer JA (1985) Reproductive biology of the hawksbill turtle *Eretmochelys imbricata* at Tortuguero, Costa Rica, with notes on the ecology of the species in the Caribbean. Biological Conservation, 34, 353 - 368.

Blamires SJ, Guinea ML and Prince RIT (2003) Influence of Nest Site Selection on Predation of flatback sea turtle (*Natator depressus*) eggs by varanid lizards in Northern Australia. Chelonian Conservation and Biology, 4, 557 - 563.

Blanco GS, Morreale SJ, Bailey H, Seminoff JA, Paladino FV and Spotila JR (2012) Post-nesting movements and feeding grounds of a resident East Pacific green turtle *Chelonia mydas* population from Costa Rica. Endangered Species Research, 18, 233 - 245.

Block BA, Jonsen ID, Jorgensen SJ, Winship AJ, Shaffer SA, Bograd SJ, Hazen EL, Foley G, Breed GA, Harrison AL, Ganong JE, Swithenbank A, Castleton M, Dewar H, Mate BR, Shillinger GL, Schaefer KM, Benson SR, Weise MJ, Henry RW and Costa DP (2011) Tracking apex marine predator movements in a dynamic ocean. Nature, 475, 86 - 90.

Blumenthal JM, Solomon JL, Bell CD, Austin TJ, Ebanks-Petrie G, Coyne MS, Broderick AC and Godley BG (2006) Satellite tracking highlights the need for international cooperation in marine turtle management. Endangered Species Research, 2, 51 - 61.

Blumenthal JM, Abreu-Grobois FA, Austin TJ, Broderick AC, Bruford MW, Coyne MS Ebanks-Petrie G, Formia A, Meylan PA, Meylan AB and Godley BJ (2009) Turtle groups or turtle soup: dispersal patterns of hawksbill turtles in the Caribbean. Molecular Ecology, 18, 4841 - 4853.

Bograd SJ, Block BA, Costa DP and Godley BJ (2010) Biologging technologies: new tools for conservation. Introduction. Endangered Species Research, 10, 1 - 7.

Bolten AB (1999) Techniques for measurement of sea turtles. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donelly M (eds). Research and Management Techniques for the Conservation of Sea Turtles. IUCN/Species Survival Commission Marine Turtle Specialist Group Publication No. 4. Blanchard, USA, pp 110 - 114.

Bolten AB (2003) The loggerhead sea turtle: a most excellent fishe. In: Bolten AB and Witherington BE (eds). Loggerhead sea turtles. Smithsonian Institution Press, Washington, D.C., pp 1 – 3.

Botkin DB, Saxe H, Araújo MB, Betts R, Bradshaw RHW, Cedhagen T, Chesson P, Dawson TP, Etterson JR, Faith DP, Ferrier S, Guisan A, Hansen AS, Hilbert DW, Loehle C, Margules C, New M, Sobel MJ and Stockwell DRB (2007) Forecasting the effects of global warming on biodiversity. BioScience, 57, 227 – 236.

Botsford LW, Brumbaugh DR, Grimes C, Kellner JB, Largier J, O'Farrell MR, Ralston S, Soulanille E and Wespestad V (2009) Connectivity, sustainability, and yield: bridging the gap between conventional fisheries management and marine protected areas. Reviews in Fish Biology and Fisheries, 19, 69 - 95.

Bouchard SS and Bjorndal KA (2000) Sea turtles as biological transporters of nutrients and energy from marine to terrestrial ecosystems. Ecology, 81, 2305 – 2313.

Boulon RH, Dutton PH and McDonald DL (1996) Leatherback turtles (*Dermochelys coriacea*) on St. Croix, US Virgin Islands: fifteen years of conservation. Chelonian Conservation and Biology, 2, 141 - 147.

Bowen BW, Grants WS, Hillis-Star Z, Shaver DJ, Bjorndal KA, Bolten AB and Bass AL (2007) Mixed stock analysis reveals the migrations of juvenile hawksbill, *Eretmochelys imbricata*, in the Caribbean Sea. Molecular Ecology, 16, 49 - 60.

Bräutigam A and Eckert KL (2006) Turning the Tide: Exploitation, Trade and Management of Marine Turtles in the Lesser Antilles, Central America, Colombia and Venezuela. TRAFFIC international, Cambridge, UK.

Broderick AC, Godley BJ, Reece S and Downie JR (2000) Incubation periods and sex ratios of green turtles: highly female biased hatchling production in the eastern Mediterranean. Marine Ecology Progress Series, 202, 273 - 281.

Broderick AC, Godley BJ and Hays GC (2001) Metabolic heating and the prediction of sex ratios for green turtles (*Chelonia mydas*). Physiological and Biochemical Zoology, 74, 161 - 170.

Brooks TM, Mittermeier RA, da Fonseca GAB, Gerlach J, Hoffmann M, Lamoreux JF, Mittermeier CG, Pilgrim JD and Rodrigues ASL (2006) Global Biodiversity Conservation Priorities, 313, 58 - 61.

Brown LS, Hart CE and Broderick AC (2012) The effects of artificial incubation on olive ridley (*Lepidochelys olivacea*) hatchlings. In: Jones TT and Wallace BP (eds). Proceedings of the Thirty-first Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-631: 322p.

Buitrago J, Guada HJ and Doyle E (2008) Conservation science in developing countries: an inside perspective on the struggles in sea turtle research and conservation in Venezuela. Environmental science and policy, 11, 562 - 578.

Bull JJ (1980) Sex determination in reptiles. The Quarterly Review of Biology, 55, 3 – 21.

Burke L and Maidens J (2004) Reefs at risk in the Caribbean. Washington: World Resources Institute.

Burnham K and Anderson D (2002) Models selection and multi-model inference. Springer Verlag, Berlin.

Burt WH (1943) Territoriality and home range concepts as applied to mammals. Journal of Mammalogy, 24, 346 - 352.

Butchart SH, Walpole M, Collen B, van Strien A, Scharlemann JP, Almond RE et al. (2010) Global biodiversity: indicators of recent declines. Science, 328, 1164 - 1168.

Butchart SHM, Scharlemann PW, Evans MI, Quader S, Arico S, Arinaitwe J, Balman M, Bennun LA, Bertzky B, Besancon C, Boucher TM, Brooks TM, Burfield IJ, Burgess ND, Chan S, Clay RP, Crosby MJ, Davidson NC, De Silva N, Devenish C, Dutson GCL, Díaz Fernández DF, Fishpoo LDC, Fitzgerald C, Foster M, Heath MF, Hockings M, Hoffmann M, Knox D, Larsen FW, Lamoreux JF, Loucks C, May I, Millett J, Molloy D, Morling P, Parr M, Ricketts TH, Seddon N, Skolnik B, Stuart SN, Upgren A and Woodley S (2012) Protecting important sites for biodiversity contributes to meeting global conservation targets. PLoS ONE. doi:10.1371/journal.pone.0032529.

Butman CA and Carlton JT (1995) Marine biological diversity: Some important issues, opportunities and critical research needs. Reviews of Geophysics, 33, 1201 - 1209.

Carr A (1987) New perspectives on the pelagic stage of sea turtle development. Conservation Biology, 1, 103 – 121.

Carr MH, Neigel JE, Estes JA, Andelman S, Warner RR and Largier JL (2003) Comparing marine and terrestrial ecosystems: implications for the design of coastal marine reserves. Ecological Applications, 13, S90 - S107.

Carreras C, Godley BJ, León YM, Hawkes LA, Revuelta O, Raga JA and Tomás J (2013) Contextualising the last survivors: population structure of marine turtles in the Dominican Republic. PLoS ONE. doi:10.1371/journal.pone.0066037.

Carrillo E, Webb G and Manolis C (1999) Hawksbill turtles (*Eretmochelys imbricata*) in Cuba: an assessment of the historical harvest and its impacts. Chelonian Conservation and Biology, 3, 264 - 280.

Caut S, Guirlet E, Jouquet P and Girondot M (2006) Influence of nest location and yolkless eggs on the hatching success of leatherback turtle clutches in French Guiana. Canadian Journal of Zoology, 84, 908 – 916.

Caut S, Guirlet E and Girondot M (2010) Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. Marine Environmental Research, 69, 254 - 261.

CBD Secretariat of the Convention on Biological Diversity (2010) Global Biodiversity Outlook 3, Montréal, 94 pages.

Chan EH (1989) White spot development, incubation and hatching success of leatherback turtle (*Dermochelys coriacea*) eggs from Rantau Abang, Malaysia. Copeia, 1, 42 - 47.

Chan EH and Solomon SE (1989) The structure and function of the eggshell of the leatherback turtle (*Dermochelys coriacea*) from Malaysia, with notes on attached fungal forms. Animal Technology, 40, 91 - 102.

Chan EH and Liew HC (1996) Decline of the leatherback population in Terengganu, Malaysia, 1956 - 1995. Chelonian Conservation and Biology, 2, 196 - 203.

Chantrapornsyl S (1992) Artificial incubation and embryonic development of olive ridley turtle eggs. Phuket Marine Biological Center Research Bulletin, 57, 41 - 50.

Cheung WWL, Lam VWY, Sarmiento JL, Kearney K, Watson R, Zeller D and Pauly D (2010) Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. Global Change Biology, 16, 24 - 35.

Chiappone M, Sluka R and Sullivan Sealey K (2000) Groupers (Pisces: Serranidae) in fished and protected areas of the Florida Keys, Bahamas and northern Caribbean. Marine Ecology Progress Series, 198, 261 - 272.

Choi GY and Eckert KL (2009) Manual of best practices for safeguarding sea turtle nesting beaches. Wider Caribbean Sea Turtle Conservation Network (WIDECAST) Technical Report No. 9. Ballwin, Missouri. 86 pp.

Cole M, Lindeque P, Halsband C and Galloway TS (2011) Microplastics as contaminants in the marine environment: A review. Marine Pollution Bulletin, 62, 2588 - 2597.

Conrad JR, Wyneken J, Garner JA and Garner S (2011) Experimental study of dune vegetation impact and control on leatherback sea turtle *Dermochelys coriacea* nests. Endangered Species Research, 15, 13 - 27.

Costa DP, Breed GA and Robinson PW (2012) New Insights into Pelagic Migrations: Implications for ecology and conservation. Annual Review of Ecology, Evolution, and Systematics, 43, 73 - 96.

Costello C, Rassweiler A, Siegel D, De Leo G, Micheli F and Rosenburg A (2010) The value of spatial information in MPA network design. Proceedings of the National Academy of Sciences, 107, 18294 - 18299.

Coyne MS and Godley BJ (2005) Satellite Tracking and Analysis Tool (STAT): an integrated system for archiving, analyzing and mapping animal tracking data. Marine Ecology Progress Series, 301, 1 - 7.

Cuevas E, Abreu-Grobois FA, Guzman-Hernandez V, Liceaga-Correa MA and van Dam RP (2008) Post-nesting migratory movements of hawksbill turtles *Eretmochelys imbricata* in waters adjacent to the Yucatan Peninsula, Mexico. Endangered Species Research, 10, 123 - 133.

Dal Pont Morisso E and Krause L (2004) How nest translocation-time, clutch size and presence of yolkless eggs affected hatching success in *Dermochelys coriacea* (Linnaeus, 1766) (testudines: dermochelyidae), at projeto Tamar-Ibama, Espírito Santo, Brazil. Cuadernos de Herpetología, 18, 59 - 64.

Davenport J and Davenport JL (2006) The impact of tourism and personal leisure transport on coastal environments: A review. Estuarine, Coastal and Shelf Science, 67, 280 - 292.

DeGregorio BA and Southwood Williard A (2011) Incubation temperatures and metabolic heating of relocated and in situ loggerhead sea turtle (*Caretta caretta*) nests at a northern rookery. Chelonian Conservation and Biology, 10, 54 - 61.

Derraik JGB (2002) The pollution of the marine environment by plastic debris: a review. Marine Pollution Bulletin, 44, 842 - 852.

Diez CE and van Dam RP (2007) Mona and Monito Island—Puerto Rico Hawksbill Turtle Research Project: Research Report for 2006. Unpublished technical report. Mona Island Hawksbill Research Group in collaboration with Departamento de Recursos Naturales y Ambientales de Puerto Rico, Chelonia, Inc., and Universidad de Puerto Rico, Rio Piedras, Puerto Rico.

Ditmer MA and Stapleton SP (2012) Factors affecting hatch success of hawksbill sea turtles on Long Island, Antigua, West Indies. PLoSONE. doi:10.1371/journal.pone.0038472.

Dodd K (1988) Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). U.S. fish wildl. service. Biological Report, pp 110.

Dominguez TJ and Villalba AA (1994) Trade of hawksbill carapaces in Santo Domingo, Dominican Republic. In: Bjorndal KA, Bolten AB, Johnson DA and Eliazar PJ (eds). Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation, Hilton Head, South Carolina, USA, pp 34 - 35.

Dominici GO (1996) Monitoreo de anidamiento de tortuga tinglar *Dermochelys coriacea* en playas del Parque Nacional Jaragua. In: Memorias del II congreso de la Biodiversidad Caribeña, Santo Domingo, Dominican Republic.

Dow WE and Eckert KL (2011) Sea turtle nesting habitat in the Wider Caribbean Region. Endangered Species Research, 15, 129 - 141.

Dow W, Eckert K, Palmer M and Kramer P (2007) An Atlas of Sea Turtle Nesting Habitat for the Wider Caribbean Region. WIDECAST Technical Report No. 6. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, Beauford, USA.

Doyle TK, Houghton JD, Suilleabhain PF, Hobson VJ, Marnell F, Davenport J and Hays GC (2008) Leatherback turtles satellite-tagged in European waters. Endangered Species Research, 4, 23 - 31.

Dulvy NK, Sadovy Y and Reynolds JD (2003) Extiction vulnerability in marine populations. Fish and Fisheries, 4, 25 - 64.

Dutton PH and McDonald D (1995) Hatch rates of leatherback (*Dermochelys coriacea*) clutches reburied with and without yolkless eggs. Proceedings of the 12th Annual Symposium on Sea Turtle Biology and Conservation.U.S. Dept. Commerce. NOAA Tech. Memo. NMFS-SEFSC-361, Jekyll Island, Georgia, EEUU, pp 177 - 178.

Dutton PH, Whitmore CP and Mrosovsky N (1985) Masculinisation of leatherback turtle *Dermochelys coriacea* hatchlings from eggs incubated in Styrofoam boxes. Biological Conservation, 31, 249 - 264.

Dutton DL, Dutton PH, Chaloupka M and Boulon RH (2005) Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biological Conservation, 126, 186 - 194.

Eckert KL (1987) Environmental unpredictability and leatherback sea turtle (*Dermochelys coriacea*) nest loss. Herpetologica, 43, 315 - 323.

Eckert KL (1995) Anthropogenic threats to sea turtles. In: Bjorndal KA (ed). Biology and Conservation of Sea Turtles. Revised edition. Smithsonian Institute Press, Washington D.C. pp 611 - 612.

Eckert KL (1999) Designing a conservation program. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ SSC Marine Turtle Specialist Group Publication No. 4, pp 6 - 8.

Eckert KL and Eckert SA (1990) Embryo mortality and hatch success in in situ and translocated leatherback sea turtle *Dermochelys coriacea* eggs. Biological Conservation, 53, 37 - 46.

Eckert KL, Wallace BP, Frazier JG, Eckert SA and Pritchard PCH (2012) Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication BTP-R4015-2012, Washington, D.C.

Eckert SA (2002) Distribution of juvenile leatherback sea turtle *Dermochelys coriacea* sightings. Marine Ecology Progress Series, 230, 289 - 293.

Eckert SA, Bagley D, Kubis S, Ehrhart L, Johnson C, Stewart K and DeFreese D (2006) Internesting, post-nesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chelonian Conservation and Biology, 5, 239 - 248.

Edgar GJ (2011) Does the global network of marine protected areas provide an adequate safety net for marine biodiversity? Aquatic Conservation: Marine and Freshwater Ecosystems, 21, 313 - 316.

Ehrhart LM, Bagley DA and Redfoot WE (2003) Loggerhead turtles in the Atlantic Ocean: geographic distribution, abundance, and population status. In: Bolten AB and Witherington BE (eds). Loggerhead Sea Turtles. Smithsonian Institution, Washington, DC, USA, pp 157 - 174.

Ervin J (2003) Protected area assessments in perspective. Bioscience, 53, 819 - 822.

ESRI (2010) ArcGIS 10.0 GIS. Redlands, CA.

Feliz P, León YM, Tomás J, Hierro K, Mateo A, Méndez MD and Raga JA (2010) Tortoiseshell trade in Santo Domingo, Dominican Republic: discouraging news for Caribbean hawksbill. In: Dean K and López Castro MC (eds). Proceedings of the Twenty-Eighth Annual Workshop on Sea Turtle Biology and Conservation, National Oceanic and Atmospheric Administration, Miami, USA, pp 247.

Ferraroli S, Georges JY, Gaspar P and Le Maho Y (2004) Where leatherback turtles meet fisheries. Nature, 429, 521 - 522.

Fish MR, Cote IM, Horrocks JA, Mulligan B, Watkinson AR and Jones AP (2008) Construction setback regulations and sea-level rise: mitigating sea turtle nesting beach loss. Ocean and Coastal Management, 51, 330 - 341.

Fitzsimmons NN, Goldizen AR, Norman JAM, Miller JD and Moritz C (1997) Philopatry of male marine turtles inferred from mitochondrial markers. Proceedings of the National Academy of Sciences, 94, 8912 - 8917.

Fleming EH (2001) Swimming against the tide: recent surveys of exploitation, trade, and management of marine turtles in the Northern Caribbean. TRAFFIC North America, Washington, DC, USA.

Foley AM, Peck SA, Harman GR and Richardson LW (2000) Loggerhead turtle (*Caretta caretta*) nesting habitat on low-relief mangrove islands in southwest Florida and consequences to hatchling sex ratios. Herpetologica, 56, 433 - 445.

Frazier J (2002) Marine turtles and international instruments: the agony and the ecstasy. Journal of International Wildlife Law and Policy, 5, 1 - 10.

Frick MG (2013) Clarification on the nomenclatural availability of the turtle barnacle genus *Chelolepas* (Cirripedia: Balanomorpha: Coronuloidea), a corrigendum to Hayashi (2012). Marine Biological Association of the United Kingdom, 93, 1579 – 1580.

Gaos AR, Lewison RL, Wallace BP, Yanez IL, Liles MJ, Nichols WJ, Baquero A, Hasbun CR, Vasquez M, Urtega J and Seminoff JA (2012) Spatial ecology of critically endangered hawksbill turtles *Eretmochelys imbricata* implications for management and conservation. Marine Ecology Progress Series, 450, 184 - 194.

García R and Roersch C (1996) Politica de manejo y utilización de los recursos florísticos en la República Dominicana. Journal of Ethnopharmacology, 51, 147 - 160.

García A, Ceballos G and Adaya R (2003) Intensive beach management as an improved sea turtle conservation strategy in Mexico. Biological Conservation, 111, 253 - 261.

Garland K and Hillis-Starr Z (2003) Carapace characteristics of hawksbills (*Eretmochelys imbricata*) at Buck Island National Monument, US Virgin Islands: long term remigrants vs neophytes. In: Seminoff JA (ed). Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. National Oceanic and Atmospheric Administration, Miami, USA, pp 286.

Garret K, Wallace BP, Garner J and Paladino FV (2010) Variations in leatherback turtle nest environments: consequences for hatching success. Endangered Species Research, 11, 147 - 155.

Gell FR and Roberts CM (2002) The fishery effects of marine reserves and fishery closures. Washington, DC 20037, USA.

Georges JY, Fossette S, Billes A, Ferraroli S, Fretey J, Grémillet D, Le Maho Y, Myers AE, Tanaka H and Hays GC (2007) Meta-analysis of movements in Atlantic leatherback turtles during the nesting season: conservation implications. Marine Ecology Progress Series, 338, 225 - 232.

Geraldes FX (2003) The coral reefs of the Dominican Republic. In Cortés J (ed) Latin American Coral Reefs. Elsevier, Amsterdam, Holanda, pp 303 - 330.

Gerber LR, Heppell SS, Ballantyne F and Sala E (2005) The role of dispersal and demography in determining the efficacy of marine reserves. Canadian Journal of Fisheries and Aquatic Sciences, 62, 863 - 87.

Godfrey MH (1997) Sex ratios of sea turtle hatchlings: direct and indirect estimates. Phd Thesis. University of Toronto.

Godfrey MH and Mrosovsky N (1997) Estimating the time between hatching of sea turtles and their emergence from the nest. Chelonian Conservation and Biology, 2, 581 - 585.

Godfrey MH and Mrosovsky N (1999) Estimating hatchling sex ratios. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 136 - 138.

Godley BJ, Broderick AC and Mrosovsky N (2001). Estimating hatchling sex ratios of loggerhead turtles in Cyprus from incubation durations. Marine Ecology Progress Series, 210, 195 - 201.

Godley BJ, Blumenthal JM, Broderick AC, Coyne MS, Godfrey MH, Hawkes LA and Witt MJ (2008) Satellite tracking of sea turtles: where have we been and where do we go next? Endangered Species Research, 4, 3 - 22.

Grandoit J (2005) Tourism as a development tool in the Caribbean and the environmental by-products: the stresses on small island resources and viable Remedies. Journal of development and social transformation 2. Available from http://www.maxwell.syr.edu/moynihan/programs/dev/journal.htlm (accessed January 19, 2013).

Gray JS (1997) Marine biodiversity: patterns, threats and conservation needs. Biodiversity and Conservation, 6, 153 – 175.

Griffiths RA and Pavajeau L (2008) Captive breeding, reintroduction, and the conservation of amphibians. Conservation Biology, 22, 852 – 61.

Groombridge B and Luxmoore R (1989) The Green Turtle and Hawksbill: World Status, Exploitation and Trade. (Secretariat of CITES, Lausanne, Switzerland) pp 601.

Guarderas AP, Hacker SD and Lubchenco J (2008) Current status of marine protected areas in Latin America and the Caribbean. Conservation Biology, 22, 1630 - 1640.

Guidetti P, Notarbartolo-Di-Sciara G and Agardy T (2013) Integrating pelagic and coastal MPAs into large-scale ecosystem-wide management. Aquatic Conservation: Marine and Freshwater Ecosystems, 23, 179 - 182.

Halpern BS (2014) Conservation: Making marine protected areas work. Nature, 506, 167 - 168.

Halpern BS, Selkoe KA, Micheli F and Kappel CV (2007) Evaluating and Ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology, 21, 1301 - 1315.

Hamman M, Godfrey MH, Seminoff JA, Arthur K, Barata PCR, Bjorndal KA, Bolten AB, Broderick AC, Campbell LM, Carreras C, Casale P, Chaloupka M, Chan SKF, Coyne MS, Crowder LB, Diez CE, Dutton PH, Epperly SP, FitzSimmons NN, Formia A, Girondot M, Hays GC, Cheng IJ, Kaska Y, Lewison R, Mortimer JA, Nichols WJ, Reina RD, Shanker K, Spotila JR, Tomás J, Wallace BP, Work TM, Zbinden J and Godley BJ (2010) Global research priorities for sea turtles: informing management and conservation in the 21st century. Endangered Species Research, 11, 245 - 269.

Hammerschlag N, Gallagher AJ and Lazarre DM (2011) A review of shark satellite tagging studies. Journal of Experimental Marine Biology and Ecology, 398, 1 - 8.

Harmelin-Vivien M, Le Diréach L, Bayle-Sempere J, Charbonnel E, García-harton JA, Ody D, Pérez-Ruzafa A, Reñones O, Sánchez Jerez P and Valle C (2008) Gradients of abundance and biomass across reserve boundaries in six Mediterranean marine protected areas: Evidence of fish spillover? Biological Conservation, 141, 1829 - 1839.

Hart KM and Hyrenbach KD (2010) Satellite telemetry of marine megavertebrates: The coming of age of an experimental science. Endangered Species Research, 10, 9 - 20.

Hart KM, Zawada DG, Fujisaki I and Lidz BH (2010) Inter-nesting habitat-use patterns of loggerhead sea turtles: enhancing satellite tracking with benthic mapping. Aquatic Biology, 11, 77 - 90.

Hart KM, Lamont MM, Sartain AR, Fujisaki I and Stephens BS (2013) Movements and habitat-use of loggerhead sea turtles in the Northern Gulf of Mexico during the reproductive period. PLoS ONE. doi:10.1371/journal.pone.0066921.

Hatase H, Matsuzawa Y, Sakamoto W, Baba N and Miyawaki I (2002) Pelagic habitat use of an adult Japanese male loggerhead turtle *Caretta caretta* examined by the Argos satellite system. Fisheries Science, 68, 945 - 947.

Hawkes LA, Broderick AC, Coyne MS, Godfrey MH, Lopez-Jurado LF, Lopez-Suarez P, Merino SE, Varo-Cruz N and Godley BJ (2006) Phenotypically linked dichotomy in sea turtle foraging requires multiple conservation approaches. Current Biology, 16, 990 - 995.

Hawkes LA, Broderick AC, Godfrey MH and Godley BJ (2009) Climate change and marine turtles. Endangered Species Research, 7, 137 - 154.

Hawkes LA, Tomás J, Revuelta O, León YM, Blumenthal JM, Broderick AC, Fish M, Raga JA, Witt MJ and Godley BJ (2012) Migratory patterns in hawksbill turtles described by satellite tracking. Marine Ecology Progress Series, 461, 223 - 232.

Hays GC, Houghton JDR and Myers AE (2004) Pan-Atlantic leatherback turtle movements. Nature, 429, 522.

Hays GC, Hobson VJ, Metcalfe JD, Righton D and Sims DW (2006) Flexible foraging movements of leatherback turtles across the North Atlantic Ocean. Ecology, 87, 2647 - 2656.

Heaslip SG, Iverson SJ, Bowen WD and James MC (2012) Jellyfish Support High energy intake of leatherback sea turtles (*Dermochelys coriacea*): video evidence from animal-Borne Cameras. PLoSONE. doi:10.1371/journal.pone.0033259.

Hendriks IE, Duarte CM and Heip CHR (2006) Biodiversity research still grounded. Science, 312, 1715.

Heredia F (2003) Manejo integrado costero marino en la República Dominicana. Publicaciones de la SEMARN.

Hillis ZM and Phillips B (1996) The year of expanding, standardizing and planning for the future of hawksbill turtle research at Buck Island reef NM. In: Proceedings of the 15th Annual Symposium on Sea Turtle Biology and Conservation, 20-25 February 1995. Hilton Head, South Carolina, USA.

Hilterman ML and Goverse E (2007) Nesting and nest success of the leatherback turtle (*Dermochelys coriacea*) in Suriname, 1999-2005. Chelonian Conservation and Biology, 6, 87 - 100.

Hirayama R (1998) Oldest known sea turtle. Nature, 392, 705 - 708.

Hirth HF (1971) Synopsis of biological data on the green turtle, *Chelonia mydas* (Linnaeus) 1758. FAO Fisheries Synopsis No 85, 1.1 - 8.19.

Hirth HF (1997) Synopsis of the Biological Data on the Green Turtle, *Chelonia mydas* (Linnaeus 1758). Biological Report 97, 1 - 129. U. S. Department of Interior.

Hoffmann M, Hilton Taylor C, Angulo A, Böhm M, Brooks TM, Butchart SHM et al (2010) The impact and shortfall of conservation on the status of the world's vertebrates. Science, 330, 1503 - 1509.

Hoffman M, Belant JL, Chanson JS, Cox NA, Lamoreux J, Rodrigues ASL, Schipper J and Stuart SN (2011) The changing fates of the world's mammals. Philosophical Transactions of the Royal Society B, 366, 2598 - 2610.

Hooge PN, Eichenlaub WM and Solomon EK (1999) The animal movement program. USGS, Alaska Biological Science Center. http://www.absc.usgs. gov/glba/gistools.

Hopkins-Murphy SR, Owens DW and Murphy TM (2003). Ecology of immature loggerheads on foraging grounds and adults in interesting habitat in the eastern United States. In: Bolten A and Witherington B (ed). Loggerhead Sea Turtles. Washington, DC: Smithsonian Institution Press, pp 79 - 92.

Horrocks JA and Scott NM (1991) Nest site location and nest success in the hawksbill turtle *Eretmochelys imbricata* in Barbados, West Indies. Marine Ecology Progress Series, 69, 1 - 8.

Horrocks JA, Krueger BH, Fastigi M, Pemberton EL and Eckert KL (2011) International movements of adult female hawksbill turtles (*Eretmochelys imbricata*): first results from the Caribbean's marine turtle tagging centre. Chelonian Conservation and Biology, 10, 18 - 25.

Houghton JDR, Doyle TK, Wilson MW, Davenport J and Hays GC (2006) Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology, 87, 1967 - 1972.

Houghton JDR, Myers AE, Lloyd C, King RS, Isaacs C and Hays GC (2007) Protracted rainfall decreases temperature within leatherback turtle (*Dermochelys coriacea*) clutches in Grenada, West Indies: Ecological implications for a species displaying temperature dependent sex determination. Journal of Experimental Marine Biology and Ecology, 345, 71 - 77.

Houghton JDR, Cedras A, Myers AE, Liebsch N, Metcalfe JD, Mortimer JA and Hays GC (2008) Measuring the state of consciousness in a free-living diving sea turtle. Journal of Experimental Marine Biology and Ecology, 356, 115 - 120.

Hulin V, Delmas V, Girondot M, Godfrey MH and Guillon JM (2009) Temperature-dependent sex determination and global change: Are some species at greater risk? Oecologia, 160, 493 - 506.

IUCN/UNEP/WWF (1980) World conservation strategy. Gland Switzerland: IUCN/UNEP/WWF.

IUCN (2011) Red list of threatened species. Available at URL: http://www.iucnredlist.org (accessed 19 November 2011).

IUCN (2012) The IUCN red list of threatened species (WWW document). URL: http://www.iucnredlist.org (accessed 19 November 2012).

IUCN (2013) Red list of threatened species (WWW document). URL: http://www.iucnredlist.org (accessed 15 May 2013).

Jackson JBC (1997) Reefs since Columbus. Coral Reefs, 16, S23 - S33.

Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Bostford LW, Bourque BJ, Bradbury RH, Cooke R, Erlandson J, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ and Warner RR (2001) Historical overfishing and the recent collapse of coastal ecosystems. Science, 293, 629 - 637.

James MC, Ottensmeyer CA and Myers RA (2005) Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. Ecology Letters, 8, 195 - 201.

James MC, Sherrill-Mix SA and Myers RA (2007) Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series, 337, 245 - 254.

Kamel S and Mrosovsky N (2005) Repeatability of nesting preferences in the hawksbill sea turtle, (*Eretmochelys imbricata*), and their fitness consequences. Animal Behaviour, 70, 819 - 828.

Kamel SJ and Mrosovsky N (2006) Deforestation: risk of sex ratio distortion in hawksbill sea turtles. Ecological Applications, 16, 923 - 931.

Kamel SJ and Delcroix E (2009) Nesting ecology of the hawksbill turtle, *Eretmochelys imbricata*, in Guadeloupe, French West Indies from 2000-07. Journal of Herpetology, 43, 367 - 376.

Katselidis KA, Schofield G, Stamou GP, Dimopoulos P and Pantis JD (2012) Females first? Past, present and future variability in offspring sex ratio at a temperate sea turtle breeding area. Animal Conservation, 15, 508 - 18.

Kerchner C, Honzák M, Kemkes R, Richardson A, Townsend J and Rimmer CC (2010) Designing spatially explicit incentive programs for habitat conservation: A case study of the Bicknell's thrush wintering grounds. Ecological Economics, 69, 2108 - 2115.

Kornaraki E, Matossiana DA, Mazarisa AD, Matsinosa YG and Margaritoulis D (2006) Effectiveness of different conservation measures for loggerhead sea turtle (*Caretta caretta*) nests at Zakynthos Island, Greece. Biological Conservation, 130, 324 - 330.

Lagueux CJ and Campbell CL (2005) Marine turtle nesting and conservation needs on the south-east coast of Nicaragua. Oryx, 39, 398 - 405.

Lagueux CJ, Campbell CL and McCoy WA (2003) Nesting and conservation of the hawksbill turtle, *Eretmochelys imbricata*, in the Pearl Cays, Nicaragua. Chelonian Conservation and Biology, 4, 588 - 602.

Larsen FW, Turner WR and Brooks TM (2012) Conserving critical sites for biodiversity provide disproportionate benefits to people. PLoSONE. doi:10.1371/journal.pone.0036971.

Laurance WF (2007) Have we overstated the tropical biodiversity crisis? Trends in Ecology and Evolution, 22, 65 - 70.

Laver PN and Kelly MJ (2008) A Critical Review of Home Range Studies. Journal of Wildlife Management, 72, 290 - 298.

Leighton PA, Horrocks JA, Krueger BH, Beggs JA and Kramer DL (2008) Predicting species interactions from edge responses: mongoose predation on hawksbill sea turtle nests in fragmented beach habitat. Proceedings of the Royal Society B: Biological Sciences, 275, 2465 - 2472.

León YM (2004) Community Impacts of Coastal Tourism in the Dominican Republic. Ph.D. Dissertation. University of Rhode Island, Kingston, Rhode Island, USA.

León YM (2007) The impact of tourism on rural livelihoods in the Dominican Republic's coastal areas. Journal of Development Studies, 43, 340 - 359.

León YM and Diez CE (1999) Population structure of hawksbill turtles on a foraging ground in the Dominican Republic. Chelonian Conservation and Biology, 3, 230 - 236.

León YM and Bjorndal KA (2002) Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. Marine Ecology Progress Series, 245, 249 - 258.

Leslie H, Ruckelshaus M, Ball IR, Andelman S and Possingham HP (2003) Using siting algorithms in the design of marine reserve networks. Ecological Applications, 13, S185 - S198.

Lewis M, Pryor R and Wilking L (2011) Fate and effects of anthropogenic chemicals in mangrove ecosystems: a review. Environmental Pollution, 159, 2328 - 2346.

Lewison RL, Crowder LB, Read AJ and Freeman SA (2004) Understanding impacts of fisheries bycatch on marine megafauna. Trends in Ecology and Evolution, 19, 598 - 604.

Liles MJ, Jandres MV, López WA, Mariona GI, Hasbún CR and Seminoff JA (2011) Hawksbill turtles *Eretmochelys imbricata* in El Salvador: nesting distribution and mortality at the largest remaining nesting aggregation in the eastern Pacific Ocean. Endangered Species Research, 14, 23 - 30.

Limpus CJ (2007) A biological review of Australian marine turtle species: 5. Flatback turtle, *Natator depressus* (Garman). Environmental Protection Agency, Queensland, 54pp.

Limpus CJ, Miller J, Parmenter C and Limpus D (2003) The green turtle, *Chelonia mydas*, population of Raine Island and the northern Great Barrier Reef 1843-2001. Memoirs of the Queensland Museum, 49, 349 - 440.

López-Mendilaharsu M, Rocha CFD, Miller P, Domingo A and Prosdocim L (2009) Insights on leatherback turtle movements and high use areas in the Southwest Atlantic Ocean. Journal of Experimental Marine Biology and Ecology, 378, 31 - 39.

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH, Cooke RG, Kay MC, Kidwell SM, Kirby MX, Peterson CH and Jackson JBC (2006) Depletion, degradation, and recovery potential of estuaries and coastal seas. Science, 312, 1806 - 1809.

Lubchenco J, Palumbi SR, Gaines SD and Andelman S (2003) Plugging a hole in the ocean: the emerging science of marine reserves. Ecological Applications, 13, S3 - S7.

Luschi P, Hays GC, Del Seppia C, Marsh R and Papi F (1998) The navigational feats of green sea turtles migrating from Ascension Island investigated by satellite telemetry. Procedings of the Royal Society B: Biological Sciences, 265, 2279 - 2284.

Macleod DVL (2001) Parks or people? National parks and the case of Del Este, Dominican Republic. Progress in Development Studies, 1, 221 - 235.

Mansfield KL, Wyneken J, Porter WP, Luo J (2014) First satellite tracks of neonate sea turtles redefine the 'lost years' oceanic niche. Proceedings of the Royal Society B: Biological Sciences, 281, 20133039.

Marcovaldi MA, Lopez G, Soaresi LS, Santos A, Bellini C and Barata P (2007) Fifteen years of hawksbill sea turtle (*Eretmochelys imbricata*) nesting in northern Brazil. Chelonian Conservation and Biology, 6, 223 - 228.

Marcovaldi MA, Lopez GG, Soares LS and Mendilaharsu M (2012) Satellite tracking of hawksbill turtles *Eretmochelys imbricata* nesting in northern Bahia, Brazil: turtle movements and foraging destinations. Endangered Species Research, 17, 123 - 132.

Maros A, Louveaux A, Godfrey MH and Girondot M (2003) *Scapteriscus didactylus* (Orthoptera, Gryllotalpidae), predator of leatherback turtle eggs in French Guiana. Marine Ecology Progress Series, 249, 289 - 296.

Martin RE (1996) Storm impacts on loggerhead turtle reproductive success. Marine Turtle Newsletter, 73, 10 - 12.

Mason TJ and Lowe CG (2010) Home range, habitat use, and site fidelity of barred sand bass within a southern California marine protected area. Fisheries Research, 106, 93 - 101.

Maulany RI, Booth DT and Baxter GS (2012) Emergence success and sex ratio of natural and relocated nests of olive ridley turtles from Alas Purwo National Park, East Java, Indonesia. Copeia, 4, 738 - 747.

Maxwell SM, Breed GA, Nickel BA, Makanga-Bahouna J, Pemo-Makaya E, Parnell RJ, Formia A, Ngouessono S, Godley BJ, Costa DP, Witt MJ and Coyne MS (2011) Using satellite tracking to optimize protection of long-lived marine species: olive ridley sea turtle conservation in central Africa. PLoSOne. doi.org/10.1371/journal.pone.0019905.

McClenachan L, Jackson J and Newman M (2006) Conservation implications of historic sea turtle nesting beach loss. Frontiers in Ecology and the Environment, 4, 290 - 296.

McGowan A, Broderick AC, Frett G, Gore S, Hastings M, Pickering A, Wheatley D, White J, Witt MJ and Godley BJ (2008) Down but not out: marine turtles of the British Virgin Islands. Animal Conservation, 11, 92 - 103.

Meylan AB (1988) Spongivory in hawksbill turtles: a diet of glass. Science, 239, 393 - 395.

Meylan AB (1999) Status of the hawksbill turtle (*Eretmochelys imbricata*) in the Caribbean Region. Chelonian Conservation and Biology, 3, 177 - 184.

Meylan AB (2002) Global Status Review of the hawksbill turtle (*Eretmochelys imbricata*), with an emphasis on the wider Caribbean. An Addendum for the Second CITES wider Caribbean hawksbill turtle dialogue meeting, Cayman Islands (U.K.) 21-23 May 2002. In Marine Turtle Specialist Group 2007 IUCN red list status assessment.

Meylan A and Meylan P (1999) Introduction to the evolution, life history, and biology of sea turtles. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 3 - 5.

Meylan A and Redlow A (2006) *Eretmochelys imbricata* -Hawksbill turtle. In: Meylan PA (ed). Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3, pp 105 - 127.

Meylan A, Schroeder B and Mosier A (1995) Sea turtle nesting activity in the State of Florida 1979-1992. Florida Marine Research Institute Publications, 52, 1 - 16.

Meylan PA, Meylan AB and Gray JA (2011) The ecology and migrations of sea turtles. 8. Tests of the developmental habitat hypothesis. Bulletin of the American Museum of Natural History, 357, 1 - 70.

Micheli F, Shelton AO, Bushinsky SM, Chiu AL, Haupt, AJ, Heiman KW, Kappel CV, Lynch MC, Martone RG, Dunbar RB and Watanabe J (2008) Persistence of depleted abalones in marine reserves of central California. Biological Conservation, 141, 1078 - 1090.

Miller JD (1997) Reproduction in sea turtles. In: Lutz PL and Musick JA (eds). The biology of marine turtles. CRC Press, Washington DC, pp 51 - 81.

Miller JD (1999) Determining clutch size and hatch success. In: Eckert KL, Bjorndal, KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ Species Survival Commission Marine Turtle Specialist Group Publication No. 4, Blanchard, USA, pp 124 - 129.

Miloslavich P, Díaz JM, Klein E, Alvarado JJ, Díaz C et al. (2010) Marine Biodiversity in the Caribbean: Regional Estimates and Distribution Patterns. PLoSONE. doi:10.1371/journal.pone.0011916.

Moncada F, Carrillo E, Saenz A and Nodarse G (1999) Reproduction and nesting of the hawksbill turtle, *Eretmochelys imbricata*, in the Cuban Archipelago. Chelonian Conservation and Biology, 3, 257 - 263.

Moncada FG, Hawkes LA, Fish MR, Godley BJ, Manolis SC, Medina Y, Nodarse G and Webb GJW (2012) Patterns of dispersal of hawksbill turtles from the Cuban shelf inform scale of conservation and management. Biological Conservation, 148, 191 - 199.

Morales M (2012) With code developed by the R Development Core Team and with general advice from the R-help listerv community, especially Duncan Murdoch. Sciplot: Scientific Graphing Functions for Factorial Designs. Rpackage version 1.1 - 0.

Morreale SJ, Ruiz GJ, Spotila JR and Standora EA (1982) Temperature-dependent sex determination: current practices threaten conservation of sea turtles. Science, 216, 1245 - 1247.

Mortimer JA (1999) Reducing threats to eggs and hatchlings: hatcheries. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp 175 - 178.

Mortimer JA and Carr A (1987) Reproduction and migrations of the Ascension Island green turtle (*Chelonia mydas*). Copeia, 103 - 113.

Mota JM and León YM (2003) Beliefs and perceptions associated with sea turtle products in the Dominican Republic. In: Pilcher NJ (ed). Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation, Kuala Lumpur, Malaysia, pp 197 - 199.

Mrosovsky N (1982) Sex ratio bias in hatchling sea turtles from artificially incubated eggs. Biological Conservation, 23, 309 - 314.

Mrosovsky N (1983) Ecology and nest-site selection of leatherback turtles, *Dermochelys coriacea*. Biological Conservation, 26, 47 - 56.

Mrosovsky N (1994) Sex ratios of sea turtles. Journal of Experimental Zoology, 270, 16 - 27.

Mrosovsky N and Pieau C (1991) Transitional range of temperature, pivotal temperatures and thermosensitive stages for sex determination in reptiles. Amphibia-Reptilia, 12, 169 - 179.

Mrosovsky, N, Baptistotte C and Godfrey MH (1999) Validation of incubation duration as an index of the ratio of hatchling sea turtles. Canadian Journal of Zoology, 77, 831 - 835.

Mrosovsky N, Kamel SJ, Diez CE and van Dam RP (2009) Methods of estimating natural sex ratios of sea turtles from incubation temperatures and laboratory data. Endangered Species Research, 8, 147 - 155.

Myers N (1988) Threatened biotas: “hot spots” in tropical forests. Environmentalist, 8, 187 - 208.

Naro-Maciel E, Mrosovsky N and Marcovaldi MA (1999). Thermal profiles of sea turtle hatcheries and nesting areas at Praia do Forte, Brazil. Chelonian Conservation and Biology, 3, 407 - 413.

Naughton-Treves L, Buck Holland M and Brandon K (2005) The role of protected areas in conserving biodiversity and sustaining local livelihoods. Annual review of Environmental resources journal, 30, 219 - 52.

Nel R, Punt AE and Hughes GR (2013) Are Coastal Protected Areas Always Effective in Achieving Population Recovery for Nesting Sea Turtles? PLoSONE. doi:10.1371/journal.pone.0063525.

Nicholls RJ, Wong PP, Burkett VR, Codignotto JO, Hay JE, McLean RF, Ragoonaden S and Woodroffe CD (2007) Coastal systems and low-lying areas. Climate change 2007: Impacts, adaptation and vulnerability. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ and Hanson CE (eds). Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK, pp 315 - 356.

Norse EA (2010) Ecosystem-based spatial planning and management of marine fisheries: Why and how? Bulletin of marine science, 86, 179 - 195.

ONU (2008) United Nations, Department of Economic and Social Affairs, Population Division (2009). World Population Prospects: The 2008 Revision, Highlights, Working Paper No. ESA/P/WP.210.

Ordoñez C, Troëng S, Meylan A, Meylan P and Ruiz A (2007) Chiriqui Beach, Panama, the most important leatherback nesting beach in Central America. Chelonian Conservation and Biology, 6, 122 - 126.

Ottenwalder JA (1981) Estudio preliminar sobre el estado, distribución, y biología reproductiva de las tortugas marinas en la República Dominicana. Tesis de Licenciatura, Universidad Autónoma de Santo Domingo, Dominican Republic.

Ottenwalder JA (1987) National Report for Dominican Republic. Western Atlantic ymposium II, Mayagüez, Puerto Rico, September 1987. WATS2-072. Unpublished.

Ottenwalder JA (1996) Conservation and management of sea turtles in the Dominican Republic. Unpublished Report.

Öz M, Erdogan A, Kaska Y, Düsen S, Aslan A, Sert H, Yavuz M and Tunc MR (2004) Nest temperatures and sex-ratio estimates of loggerhead turtles at Patara beach on the south western coast of Turkey. Canadian Journal of Zoology, 82, 94 - 101.

Özdemir B and Türkozan O (2006) Carapacial scute variation in green turtle, *Chelonia mydas* hatchlings in Northern Cyprus. Turkish Journal of Zoology, 30, 141 - 146.

Paramo J, Guillot-Illidge L, Benavides S, Rodríguez A and Sánchez-Ramírez C (2009) Aspectos poblacionales y ecológicos de peces demersales de la zona norte del caribe colombiano en relación con el hábitat: una herramienta para identificar áreas marinas protegidas (AMPs) para el manejo pesquero. Caldasia, 31, 123 - 144.

Parsons JJ (1962) The Green Turtle and Man. University of Florida Press, Gainesville, USA.

Patiño-Martinez J, Marco A, Quiñones L and Godley BJ (2008) Globally significant nesting of the leatherback turtle (*Dermochelys coriacea*) on the Caribbean coast of Colombia and Panama. Biological Conservation, 141, 1982 - 1988.

Patiño-Martinez J, Marco A, Quiñones L and Calabuig C (2010) Los huevos falsos (SAGs) facilitan el comportamiento social de emergencia en las crías de la tortuga laúd *Dermochelys coriacea* (Testudines: Dermochelyidae). Revista de Biología Tropical, 58, 943 - 954.

Patiño-Martínez J, Marco A, Quiñones L, Abella E, Muriel Abad R and Diéguez-Uribeondo J (2011) How do hatcheries influence embryonic development of sea turtle eggs? Experimental analysis and isolation of microorganisms in leatherback turtle eggs. Journal of Experimental Zoology, 313A.

Peckham SH, Diaz DM, Walli A, Ruiz G, Crowder LB and Nichols WJ (2007) Small-Scale fisheries bycatch jeopardizes endangered Pacific loggerhead turtles. PLoS ONE. doi:10.1371/journal.pone.0001041.

Penha-Lopes G, Torres P, Cannicci S, Narciso L and Paula J (2011) Monitoring anthropogenic sewage pollution on mangrove creeks in southern Mozambique: a test of Palaemon concinnus Dana, 1852 (Palaemonidae) as a biological indicator. Environmental Pollution, 159, 636 - 645.

Perdomo L, Arias Y, León YM and Wege D (2010) Áreas importantes para la conservación de las aves en la República Dominicana. Grupo Jaragua y el programa IBA-Caribe de BirdLife international. Santo Domingo, República Dominicana.

Pérez-Castañeda R, Salum-Fares A and Defeo O (2007) Reproductive patterns of the hawksbill turtle *Eretmochelys imbricata* in sandy beaches of the Yucatan Peninsula. Journal of the Marine Biological Association of the United Kingdom, 87, 815 - 824.

Perrault JR, Miller DL, Eads E, Johnson C, Merrill A, Thompson LJ and Wyneken J (2012) Maternal health status correlates with success of leatherback sea turtles (*Dermochelys coriacea*) from Florida. PLoSONE. doi:10.1371/journal.pone.0031841.

Phillott AD and Parmenter CJ (2007) Deterioration of green (*Chelonia mydas*) eggs after known embryo mortality. Chelonian Conservation and Biology, 6, 262 - 266.

Pinheiro J, Bates D, DebRoy S, Sarkar D, R Development Core Team (2011) nlme: Linear and nonlinear mixed effects models. R package version 3.1 - 102.

Pintus KJ, Godley BJ, McGowan A and Broderick AC (2009) Impact of clutch relocation on green turtle offspring. Journal of Wildlife Management, 73, 1151 - 1157.

Polidoro BA, Carpenter KE, Collins L, Duke NC, Ellison AM, Ellison JC, Farnworth EJ, Fernando ES, Kathiresan K, Koedam NE, Livingstone SR, Miyagi T, Moore GE, Nam VN, Ong JE, Primavera JH, Salmo SG III, Sanciangco JC, Sukardjo S, Wang Y and Yong YWH (2010) The loss of species: mangrove extinction risk and geographic areas of global concern. PLoS One. doi:10.1371/journal.pone.0010095.

Powell RA (2000) Animal home ranges and territories and home range estimators. In: Boitani L, Fuller T (eds), Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York, pp 65 - 110.

Primack RB (1993) Essentials of Conservation Biology. Sunderland, MA: Sinauer & Associates.

Pritchard P (1995) Recovered sea turtle populations and U.S. recovery team efforts. In: Bjorndal K (ed). Biology and conservation of sea turtles. Washington, DC: Smithsonian Institution Press, pp 503 - 511.

Pritchard PCH (1996) Evolution, phylogeny and current status. In: Lutz PL and Musick JA (eds). The Biology of Sea Turtles. CRC Press, Florida, pp 1 - 28.

Pritchard PCH and Mortimer JA (1999) Taxonomy, external orphology, and species identification. In: Eckert KL, Bjorndal KA, Abreu-Grobois FA and Donnelly M (eds). Research and Management Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publ. No. 4. Washington, D.C., pp 21 - 38.

Pritchard P, Bacon P, Berry F, Carr A, Fletmeyer J, Gallagher R, Hopkins S, Lankford R, Marquez R, Ogren ML, Pringlejr W, Reichart H and Witham R (1993) Manual sobre tecnicas de investigacion y conservacion de las sea turtles, second ed. Center for the Enviromental Education, Washington, DC.

Pullin AS and Knight TM (2003) Support for decision making in conservation practice: an evidence-based approach. Journal for Nature Conservation, 11, 83 - 90.

Pullin AS and Knight TM (2009) Doing more good than harm: building an evidence-base for conservation and environmental management. Biological Conservation, 142, 931 - 934.

Pullin AS, Knight TM, Stone DA and Charman K (2004) Do conservation managers use scientific evidence to support their decision-making? Biological Conservation, 119, 245 - 252.

Rabon DR, Johnson SA, Boettcher R, Dodd M, Lyons M, Murphy S, Ramsey S, Roff S and Stewart K (2003) Confirmed leatherback turtle (*Dermochelys coriacea*) nests from North Carolina, with a summary of leatherback nesting activities north of Florida. Marine Turtle Newsletter, 101, 4 – 8.

Rafferty AR, Santidrián Tomillo P, Spotila JR, Paladino FV and Reina RD (2011) Embryonic death is linked to maternal identity in the leatherback turtle (*Dermochelys coriacea*). PLoS ONE. doi:10.1371/journal.pone.0021038.

Read AJ, Drinker P and Northridge S (2006) Bycatch of marine mammals in U.S. and global fisheries. Conservation Biology, 20, 163 - 169.

Reaka-Kudla ML (2001) Known and unknown biodiversity, risk of extinction, and conservation strategy in the sea. In: Bendell-Young L and Gallaugher P (eds). Global Waters in Peril, Kluwer Academic Publishers, N.Y., pp 19 – 33.

Rees AF, Al Saady S, Broderick AC, Coyne MS, Papathanasopoulou N and Godley BJ (2010) Behavioural polymorphism in one of the world's largest populations of loggerhead sea turtles *Caretta caretta.* Marine Ecology Progress Series, 418, 201 - 212.

Reuter A and Allan C (2006) Tourists, Turtles and Trinkets: A Look at the Trade in Marine Turtle Products in the Dominican Republic and Colombia. TRAFFIC North America, Washington, DC, USA.

Revuelta O, León YM, Feliz P, Godley BJ, Raga JA and Tomás J (2012) Protected areas host important remnants of marine turtle nesting stocks in the Dominican Republic. Oryx, 46, 348 - 358.

Revuelta O, León YM, Aznar FJ, Raga JA and Tomás J (2013) Running against time: conservation of the remaining hawksbill turtle (*Eretmochelys imbricata*) nesting population in the Dominican Republic. Journal of the Marine Biological Association of the United Kingdom, 93, 1133 - 1140.

Richardson JI, Bell R and Richardson TH (1999) Population ecology and demographic implications drawn from an 11-year study of nesting hawksbill turtles, *Eretmochelys imbricata*, at Jumby Bay, Long Island, Antigua, West Indies. Chelonian Conservation and Biology, 3, 244 - 250.

Richardson JI, Hall DB, Mason PA, Andrews KM, Bjorkland R, Cai Y and Bell R (2006) Eighteen years of saturation tagging data reveal a significant increase in nesting hawksbill sea turtles (*Eretmochelys imbricata*) on Long Island, Antigua. Animal Conservation, 9, 302 - 307.

Richardson PB, Bruford M, Calosso MC, Campbell LM, Clerveau W, Formia A, Godley BJ, Henderson C, McClellan K, Newman S, Parsons K, Pepper M, Ranger S, Silver JJ, Slade L and Broderick AC (2009) Marine turtles in the Turks and Caicos Islands: remnant rookeries, regionally significant foraging stocks and a major turtle fishery. Chelonian Conservation and Biology, 8, 192 - 207.

Roberts C, McLean C, Veron J, Hawkins J, Allen G, McAllister DE, Mittermeier CG, Schueler FW, Spalding M, Wells F, Vynne C and Werner TB (2002) Marine biodiversity hotspots and conservation priorities for tropical reefs. Science, 295, 1280 - 1284.

Roberts CM and Hawkins JP (1999) Extinction risk in the sea. Trends in Ecology and evolution, 14, 241 - 246.

Roberts CM, Bohnsack JA, Gell F, Hawkins JP and Goodridge R (2001) Effects of marine reserves on adjacent fisheries. Science, 294, 1920 - 1923.

Robinson JG (2006) Conservation biology and real world conservation. Conservation Biology, 20, 658 - 669.

Rodríguez Demorizi E (1942) Relaciones históricas de Santo Domingo. In: Montalvo (ed). Ciudad Trujillo, RD, pp 1 - 254.

Roe JH, Sill NS, Columbia MR and Paladino FV (2011) Trace metals in eggs and hatchlings of Pacific leatherback turtles (*Dermochelys coriacea*) nesting at Playa Grande, Costa Rica. Chelonian Conservation and Biology, 10, 3 - 9.

Rogers AD and Laffoley Dd'A (2011) International Earth system expert workshop on ocean stresses and impacts. Summary report. IPSO Oxford, pp 18.

Ross JP and Ottenwalder JA (1983) The leatherback sea turtle, *Dermochelys coriacea*, nesting in the Dominican Republic. In: Rhodin AGJ and Miyata K (eds). Advances in Herpetology and Evolutionary Biology. Museum of Comparative Zoology, Harvard University, Cambridge, USA, pp 706 - 712.

Rosser AM and Mainka SA (2002) Overexploitation and species extinctions. Conservation Biology, 16, 584 - 586.

Rupp E, Incháustegui SJ and Arias Y (2005) Conservation of Cyclura ricordii in the southwestern Dominican Republic and a brief history of the Grupo Jaragua. Iguana, 12, 222 - 234.

Sale PF, Cowen RK, Danilowicz BS, Jones GP, Kritzer JP, Lindeman KC, Planes S, Polunin NVC, Russ GR, Sadovy YJ and Steneck RS (2005) Critical science gaps impede use of no-take fishery reserves. Trends in Ecology and Evolution, 20, 74 - 80.

Santidrián Tomillo P, Saba V, Piedra R, Paladino FV and Spotila JR (2008) Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. Conservation Biology, 22, 1216 - 1224.

Santidrián Tomillo P, Suss JS, Wallace BP, Magrini KD, Blanco G, Paladino FV and Spotila JR (2009) Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology, 156, 2021 - 2031.

Sarti Martínez L, Barragán AR, García Muñoz D, García N, Huerta P and Vargas F (2007) Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology, 6, 70 - 78.

Sato F and Madriasau BB (1991) Preliminary report on natural reproduction of hawksbill sea turtles in Palau. Marine Turtle Newsletter, 55, 12 - 14.

Sazima I and Grossman A (2006) Turtle riders: remoras on marine turtles in Southwest Atlantic. Neotropical Ichthyology, 4, 123 - 126.

Schall R (1991) Estimation in generalized linear models with random effects. Biometrika, 78, 719 - 727.

Schofield G, Scott R, Dimadi A, Fossette S, Katselidis KA, Koutsoubas D, Lilley MKS, Pantis JD, Karagouni AD and Hays GC (2013) Evidence-based marine protected area planning for a highly mobile endangered marine vertebrate. Biological Conservation, 161, 101 - 109.

Schroeder B and Murphy S (1999) Population surveys (ground and aerial) on nesting beaches. In: Eckert KL, Bjorndal, KA, Abreu-Grobois FA and Donnelly M (eds). Research and management techniques for the conservation of sea turtles. IUCN/ Species Survival Commission Marine Turtle Specialist Group Publication No. 4, Blanchard, USA, pp 45 - 53.

Scott R, Hodgson DJ, Witt MJ, Coyne MS, Adnyana W, Blumenthal JM, Broderick AC, Canbolat AF, Catry P, Ciccione S, Delcroix E, Hitipeuw C, Luschi P, Pet-Soede L, Pendoley K, Richardson PB, Rees AF and Godley BJ (2012) Global analysis of satellite tracking data shows that adult green turtles are significantly aggregated in Marine Protected Areas. Global ecology and biogeography, 21, 1053 - 1061.

Seminoff JA (2002) Marine Turtle Specialist Group (2002) global green turtle (*Chelonia mydas*) assessment for the IUCN Red List Programme. Unpublished report to Species Survival Commission, Gland, Switzerland, pp 93.

Seminoff JA and Shanker K (2008) Marine turtles and IUCN Red Listing: A review of the process, the pitfalls, and novel assessment approaches. Journal of Experimental Marine Biology and Ecology, 356, 52 - 68.

Smith ML, Hedges SB, Buck W, Hemphill A, Inchaustegui S, Ivie MA, Martina D, Maunder M and Ortega JF (2004) Caribbean Islands. In: Mittermeier RA, Gil PR, Hoffmann M, Pilgrim J, Brooks T, Mittermeier CG, Lamoreux J and da Fonseca GAB (eds). Hotspots Revisited: Earth's biologically richest and most endangered ecoregions. Mexico City (Mexico). CEMEX, pp112 - 118.

Sokal RR and Rohlf FJ (1995) Biometry: the principles and practice of statistics in biological research. 3rd edition. New York: W.H. Freeman and Co.

Soto JM, Beheregaray RCP and de P Rebello RAR (1997) Range extension: nesting by *Dermochelys* and *Caretta* in southern Brazil. Marine Turtle Newsletter, 77, 6 - 7.

Soulé ME and Wilcox BA (eds) (1980) Conservation Biology: An Ecological-Evolutionary Perspective. Sinauer Associates, Sunderland, MA.

Spalding M and Kramer P (2004) The Caribbean. In: Glover LK and Earle SA (eds). Defying Ocean's End: An Agenda for Action. Island Press, Washington, D.C. p 7 - 41.

Spalding M, Fox H, Allen G, Davidson N, Ferdaña Z, Finlayson M, Halpern B, Jorge M, Lombana A, Lourie S, Martin K, Mcmanus E, Molnar J, Recchia C and Robertson J (2007) Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. BioScience, 57, 573 - 583.

Spotila JR (2004) Sea Turtles: A complete guide to their biology, behavior, and conservation. Baltimore: Johns Hopkins University Press.

Spotila JR, Reina RD, Steyermark AC and Plotkin PT (2000) Pacific leatherback turtles face extinction. Nature, 405, 529 - 530.

Spurgeon JP (1992) The economic valuation of coral reefs. Marine Pollution Bulletin, 24, 529 - 536.

Stam S and Stam R (1992) Turtle trouble in the Dominican Republic. Marine Turtle Newsletter, 57, 19 - 21.

Stewart K, Johnson C and Godfrey MH (2007) The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean Basins. Herpetological Journal, 17, 123 - 128.

Stork NE (2010) Re-assessing current extinction rates. Biodiversity and Conservation, 19, 357 - 371.

Stork NE, Coddington JA, Colwell RK, Chazdon RL, Dick CW, Peres CA, Sloan S and Willis K (2009) Vulnerability and resilience of tropical forest species to land-use change. Conservation Biology, 23, 1438 - 1447.

Sumaila UR, Cheung W, Dyck A, Gueye K, Huang L, Lam V, Pauly D, Srinivasan T, Swartz W, Watson R and Zeller D (2012) Benefits of rebuilding global marine fisheries outweigh costs. PLoS ONE. doi:10.1371/journal.pone.0040542.

Sutherland WJ, Adams WM, Aronson RB, Aveling R, Blackburn TM, Broad S, Ceballos G, Coté IM, Cowling RM, Da Fonseca GAB, Dinerstein E, Ferraro PJ, Fleishman E, Gascon C, Hunter M JR, Hutton J, Kareiva P, Kuria A, Macdonald DW, Mackinnon K, Madgwick FJ, Mascia MB, McNeely J, Milner-Gulland EJ, Moon S, Morley CG, Nelson S, Osborn D, Pai M, Parsons ECM, Peck LS, Possingham H, Prior SV, Pullin AS, Rands MRW, Ranganathan J, Redford KH, Rodriguez JP, Seymour F, Sobel J, Sodhi NS, Stott A, Vance-borland K and Watkinson AR (2009) One hundred questions of importance to the conservation of global biological diversity. Conservation Biology, 23, 557 - 567.

Taylor E, Baine M, Killmer A and Howard M (2013) Seaflower marine protected area: Governance for sustainable development. Marine Policy, 41, 57 - 64.

Thayer GW, Engel DW and Bjorndal KA (1982) Evidence for short-circuiting of the detritus cycle of seagrass beds by the green turtle *Chelonia mydas*. Journal of experimental marine biology and ecology, 62, 173 - 183.

Toropova C, Meliane I, Laffoley D, Matthews E and Spalding M (eds) (2010) Global Ocean Protection: Present Status and Future Possibilities. Brest, France. Agence des aires marines protégées, Gland, Switzerland, Washington, DC and New York, USA: IUCN WCPA, Cambridge, UK: UNEP-WCMC, Arlington, USA: TNC, Tokyo, Japan: UNU, New York, USA: WCS, p 96.

Torres RE, Bustamante G, Chiappone M, Geraldes FX, Pugibet E, RodriguezY, Sealey KMS, Tschirky J and Vega M (2000) Fisheries zoning plan for Parque Nacional del Este, Dominican Republic. Proceedings of the Gulf and Caribbean Fisheries Institute, 51, 475 - 491.

Troëng S and Rankin E (2005) Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biological Conservation, 121, 111 - 116.

Troëng S, Chacón D and Dick B (2004) Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. Oryx, 38, 395 - 403.

Troëng S, Dutton PH and Evans D (2005) Migration of hawksbill turtles *Eretmochelys imbricata* from Tortuguero, Costa Rica. Ecography, 28, 394 - 402.

Troëng S, Harrison E, Evans D, De Haro A and Vargas E (2007) Leatherback turtle nesting trends and threats at Tortuguero, Costa Rica. Chelonian Conservation and Biology, 6, 117 - 122.

Tucker AD (2010) Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: implications for stock estimation. Journal of Experimental Marine Biology and Ecology, 383, 48 - 55.

Tuttle J and Rostal D (2010) Effects of nest relocation on nest temperature and embryonic development of loggerhead sea turtles (*Caretta caretta*). Chelonian Conservation and Biology, 9, 1 - 7.

van Dam RP and Diez CE (1998) Caribbean hawksbill turtle morphometrics. Bulletin of Marine Science, 62, 145 - 155.

van Dam RP, Diez CE, Balazs GH, Colón Colón LA, McMillan WO and Schroeder B (2008) Sex-specific migration patterns of hawksbill turtles breeding at Mona Island, Puerto Rico. Endangered Species Research, 4, 85 - 94.

Velez-Zuazo X, Ramos WD, van Dam RP, Diez CE, Abreu-Grobois FA and Mcmillan WO (2008) Dispersal, recruitment and migratory behaviour in a hawksbill sea turtle aggregation. Molecular Ecology, 17, 839 - 853.

Veron JEN, Hoegh-Guldberg O, Lenton TM, Lough JM, Obura DO, Pearce-Kelly P, Sheppard CRC, Spalding M, Stafford-Smith MG and Rogers AD (2009) The coral reef crisis: The critical importance of <350 ppm CO2. Marine pollution bulletin, 58, 1428 - 1436.

Walcott J, Eckert S and Horrocks JA (2012) Tracking hawksbill sea turtles (*Eretmochelys imbricata*) during inter-nesting intervals around Barbados. Marine Biology, 159, 927 - 938.

Wallace BP, Sotherland PR, Spotila JR, Reina RD, Franks BF and Paladino FV (2004) Biotic and abiotic factors affect the nest environment of embryonic leatherback turtles, *Dermochelys coriacea.* Physiological and Biochemical Zoology, 77, 423 - 432.

Wallace BP, Sotherland PR, Santidrian Tomillo P, Bouchard SS, Reina RD, Spotila JR and Paladino FV (2006) Egg components, egg size, and hatchling size in leatherback turtles. Comparative Biochemistry and Physiology, 145, 524 - 532.

Wallace BP, Sotherland PR, Tomillo PS, Reina RD, Spotila JR and Paladino FV (2007) Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia, 152, 37 - 47.

Wallace BP, Dimatteo AD, Hurley BJ, Finkbeiner EM, Bolten AB, Chaloupka MY, Hutchinson BJ, Abreu-Grobois FA, Amorocho D, Bjorndal KA, Bourjea J, Bowen BW, Briseño Dueñas R, Casale P, Choudhury BC, Costa A, Dutton PH, Fallabrino A, Girard A, Girondot M, Godfrey MH, Hamann M, López-Mendilaharsu M, Marcovaldi MA, Mortimer JA, Musick JA, Nel R, Pilcher NJ, Seminoff JA, Troëng S, Whiterington B and Mast RB (2010) Regional management units for marine turtles: framework for prioritizing conservation and research across multiple scales. PloS One. doi:10.1371/journal.pone.0015465.

Wallace BP, DiMatteo AD, Bolten AB, Chaloupka MY, Hutchinson BJ, Abreu-Grobois FA, Mortimer JA, Seminoff JA, Amorocho D, Bjorndal KA, Bourjea J, Bowen BW, Briseño Dueñas R, Casale P, Choudhury BC, Costa A, Dutton PH, Fallabrino A, Finkbeiner EM, Girard A, Girondot M, Hamann M, Hurley BJ, López-Mendilaharsu M, Marcovaldi MA, Musick JA, Nel R, Pilcher NJ, Troëng S, Witherington B and Mast RB (2011) Global Conservation Priorities for Marine Turtles. PLoS ONE. doi:10.1371/journal.pone.0024510.

Wallace BP, Kot CY, DiMatteo AD, Lee T, Crowder LB and Lewison RL (2013) Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. Ecosphere, 4, art40.

Watanabe KK, Hatase H, Kinoshita M, Omuta K, Bando T, Kamezaki N, Sato K, Matsuzawa Y, Goto K, NakashimaY, Takeshita H, Aoyama J and Tsukamoto K (2011) Population structure of the loggerhead turtle *Caretta caretta*, a large marine carnivore that exhibits alternative foraging behaviors. Marine Ecology Progress Series, 424, 273 - 287.

Weber S, Broderick AC, Groothuis TGG, Ellick J, Godley BJ and Blount JD (2011) Fine-scale thermal adaptation in a green turtle nesting population. Proceedings of the Royal Society B, 279, 1077 - 1084.

White TH Jr, Jhonson Camacho A, Bloom T, Lancho Diéguez P and Sellares R (2011) Human attitudes and perceptions regarding endangered species conservation: a case study of Saona Island, Dominican Republic. Latin American Journal of Conservation, 2, 18 - 29.

Whiting S, Hartley S, Lalara S, White D, Bara T, Maminya-munja C and Wurramarrba L (2006) Hawksbill turtle tracking as part of initial sea turtle research and conservation at Groote Eylandt, Northern Australia. Marine Turtle Newsletter, 114, 14 - 15.

Whitmore CP and Dutton PH (1985) Infertility, embryonic mortality and nest-site selection in leatherback and green sea turtles in Suriname. Biological Conservation, 34, 251 - 272.

Wibbels T (2003) Critical approaches to sex determination in sea turtles. In: Lutz PL, Musick JA and Wyneken J (eds). The biology of sea turtles. Volume II. Boca Raton, FL. CRC Press, pp 103 - 134.

Wielgus J, Cooper E, Torres R and Burke L (2010) Coastal Capital: Dominican Republic. Case studies on the economic value of coastal ecosystems in the Dominican Republic. Working paper.World Resources Institute, Washington, DC, USA.

Wilkinson CR (2000) Status of Coral Reefs of the World: 2000. Australian Institute of Marine Science, Townsville, Australia.

Wilson EO (1992) The Diversity of Life. Cambridge, MA: The Belknap Press of Harvard University Press.

Witt MJ, Hawkes LA, Godfrey MH, Godley BJ and Broderick AC (2010) Predicting the impacts of climate change on a globally distributed species: the case of the loggerhead turtle. The Journal of Experimental Biology, 213, 901 - 911.

Witt MJ, Bonguno EA, Broderick AC, Coyne MS, Formia A, Gibudi A, Mounguengui GA, Moussounda C, NSafou M, Nougessono S, Parnell RJ, Sounguet G, Verhage S and Godley BJ (2011) Tracking leatherback turtles from the world's largest rookery: assessing threats across the South Atlantic. Proceedings of the Royal Society B, 278, 2338 - 2347.

Witzell WN (1983) Synopsis of biological data on the hawksbill turtle *Eretmochelys imbricata* (Linnaeus, 1766). FAO Fisheries Synopsis No. 137. 78 pages.

Wlodarska-Kowalczuk M, Weslawski JM, Warzocha J and Janas U (2010) Habitat loss and possible effects on local species richness in a species-poor system - a case study of southern Baltic Sea macrofauna. Biodiversity and Conservation, 19, 3991 - 4002.

Wood LJ (2007) MPA Global: A Database of the World's Marine Protected Areas. Sea Around Us Project, UNEP-WCMC & WWF, Vancouver, Canada. Http://www.mpaglobal.org [accessed 5 July 2013].

Yntema CL and Mrosovsky N (1982) Critical periods and pivotal temperatures for sexual differentiation in loggerhead sea turtles. Canadian Journal of Zoology, 60, 1012 - 1016.

Zbinden JA, Aebischer A, Margaritoulis D and Arlettaz R (2007) Insights into the management of sea turtle internesting area through satellite telemetry. Biological Conservation, 137, 157 - 162.

Zbinden JA, Davy C, Margaritoulis D and Arlettaz R (2007) Large spatial variation and female bias in the estimated sex ratio of loggerhead sea turtle hatchlings of a Mediterranean rookery. Endangered Species Research, 3, 305 - 312.

Zuur AF, Leno EN and Meesters EHWG (2009) A Beginner's Guide to R. Springer, New York.

Zuur AF, Leno EN and Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods in Ecology & Evolution, 1, 3 - 14.

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THIS THESIS IS FOCUSED ON IDENTIFYING THE MAIN MARINE TURTLE NESTING ROOKERIES IN THE DOMINICAN REPUBLIC, DESCRIBING THE CURRENT SPATIO-TEMPORAL PATTERNS OF NESTING, AND ASSESSING THE LIKELY IMPACT OF THE CURRENT THREATS TO THESE NESTING STOCKS, SPANNING A 5-YEAR STUDY PERIOD (2006 - 2010).

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