The impact of artificial lights and anthropogenic noise on Loggerheads (*Caretta caretta*) and Green Turtles (*Chelonia mydas*), assessed at index nesting beaches in Turkey and Mexico

Dissertation

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Summary

Coastal development is considered to be a significant hazard that endangers sea turtles by destroying nesting space. If uncontrolled, it will result in decline and local extinction, or will prevent recovery of these endangered species. Light pollution (excess of artificial light at night) is known to affect the site choice of sea turtle females and the seaward orientation of hatchlings at nesting beaches. The results of this are decreased nesting efforts and increased hatchling mortality (WITHERINGTON, 1992a, 1997). It is expected that light pollution will further increase worldwide, with a 10% increase per year, in areas where no counter-measures are taken. Therefore, for sea turtle conservation, identification and designation of critical habitats and development of mitigation measures at nesting beaches are crucial. Light pollution maps specifying sea turtle index nesting sites on a global scale remain unavailable. This thesis investigates light pollution at sea turtle nesting sites of global importance, with a focus on the Mediterranean. To this end, the Top ten global nesting sites (nesting females/year) for two species, the Loggerhead (*Caretta caretta*) and the Green Turtle (*Chelonia mydas*), were identified from the literature and entered into a geo-database using Arc View GIS. Nesting site coordinates were intersected with light pollution maps, showing the propagation of light at sea level in 1996/1997 (CINZANO et al., 2001a). On a global scale, nesting beaches in Japan, SE- USA and the eastern Mediterranean are located within light pollution hotspots near urban areas (Figs. 3.3 - 3.6). Mediterranean nesting sites are of global importance for *C. caretta* populations and have regional importance for *C. mydas*. Newer nesting data (1990-2004), which were available for index nesting sites here, were compared with light pollution maps (Figs. 3.9 - 3.13, 3.15, 3.16). In the Mediterranean, 76% of the *C. caretta* and 79% of the *C. mydas* index nesting sites were affected by light pollution, which was a > 10% increase in natural light levels at night. Twenty-eight percent (*C. caretta*) and 43% (*C. mydas*) of the index sites were located within light pollution hotspots, which are up to 3 times brighter than natural sky brightness at night (Figs. 3.8, 3.14). However, no quantitative data on light pollution measured on the ground were available for sea turtle nesting beaches in the Mediterranean or any other index sites, apart from nesting sites in the United States. Therefore fieldwork was done at Belek, which is the most important *C. caretta* index nesting site in Turkey and one of the largest in the entire Mediterranean. Despite its importance for this species, Belek is affected by mass tourism and coastal development, also resulting in light pollution.

Within a beach section of 16.1 kilometres, I identified a minimum of 57 polychromatic Metal Halide (MH) lamps and 16 High Pressure Sodium Vapour (HPS) lamps close to the egg-laying zone (Figs. 4.11 – 4.16). These were the main contributors to increased Illuminance, mainly in front of hotels. At Belek, high resolution data on light pollution measured on the ground was used to investigate the effects on sea turtle nesting density, hatchling disorientation, and hatchling mortality. Statistical evaluation confirmed that light pollution by MH and HPS lamps affected the nest site selection of *C. caretta* females, which tried to avoid illuminated places and shifted from hotel zones to the few remaining darker, undeveloped areas. Seaward orientation of hatchlings was disturbed and disorientation was also high in darker areas due to higher nest density and light propagation from adjacent hotel zones. Hatchling disorientation and mortality were positively correlated (Table 4.20), highlighting the need for effective mitigation measures at Belek for species conservation. Because polychromatic Metal Halide lamps were identified to be the key problem at Belek, a set of hatchling orientation experiments was performed, investigating the effect of dichroic filters attached in front of a Tungsten Halogen lamp with similar spectral properties as the Metal Halide lamps in the field. These experiments were conducted in Xcacel, Mexico, because this nesting site provided better experimental conditions, less light pollution, and higher hatchling sample sizes compared with the Mediterranean site. In addition, a second species (*C. mydas*) could be included in the experiments. Each of five distinct dichroic filters tested cut off a proportion of the Tungsten Halogen lamp's emitted spectrum between 350 and 850 nm (Figs. 5.8). My experimental findings suggest that the filter cutting off short wavelengths below 520 nm elicited aversion to yellow light (xanthophobia) in *C. caretta* hatchlings, thus reducing disorientation. This mitigating effect was smaller in *C. mydas*.

The impact of sounds on sea turtle hatchlings has not been thoroughly investigated in behavioural experiments. Electrophysiological data for juvenile *C. mydas* (RIDGWAY et al., 1969) confirm that sea turtles are capable of perceiving low frequency aerial sounds. In a first step, I investigated ambient sounds at Belek to test for possible effects on sea turtle hatchlings. Surf/wave sounds and anthropogenic noise were identified to be the predominant sounds in this coastal area. Some noise sources had definite peaks in the low frequency range < 1000 Hz, similar to surf sound (Table 4.15). My sound measurements were compared with literature data on sea turtle sound perception (Fig. 4.18). This comparison revealed that sea turtles in the egg-

laying zone at 20 m from the shore are physically able to perceive low frequency aerial sounds at moderate sound pressure levels ($\pm$ 50 dB SPL re 20 µPa). Based on these outcomes I tested hatchlings' orientation behaviour under experimental conditions in Mexico. Low frequency test sounds (Figs. 5.6) were used as stimuli on *C. caretta* and *C. mydas* hatchlings. It could neither be proven that these test sounds were significantly repelling hatchlings, nor if the hatchlings were orienting towards the sounds. However, a high proportion of inactive hatchlings were observed in most sound simulation experiments except in the control groups (Fig. 5.12 a, b). This led to the conclusion that, in total darkness, sounds may cause inhibition of crawling in hatchlings. Inhibition was not observed in the experiments using lights exclusively (Fig. 5.13 a, b). I also tested the effect of wave/surf sound recordings versus light stimuli on hatchlings simultaneously. As with testing light stimuli only, I found that *C. caretta* and *C. mydas* hatchlings significantly oriented towards short visible wavelength light stimuli (< 520 nm) but both species exhibited some degree of xanthophobia (Fig. 5.14 a, b). Overall my investigation confirms that light stimuli have a predominant effect on sea turtle hatchlings compared with sounds. Wave/surf sounds probably do not provide an orientation cue to hatchlings. In contrast, indicated by the consistently high proportion of inactive hatchlings only in the sound treatment groups, the inhibition of natural behavioural patterns by artificial sounds may be possible. Based on the outcomes of my studies I drafted recommendations for the Mediterranean sea turtle index nesting site at Belek. Primarily, this is the overall reduction of light pollution. This should be achieved by abandoning new building projects within the coastal zone and by rebuilding light barriers such as dunes and vegetation. Further proposals are the regulation of night-time lighting at hotels, the replacement of polychromatic lights and the application of colour filters, which I found adequate to reduce hatchling disorientation. Reducing high-intensity noise after nightfall, in particular the regulation of traffic and relocation of dance-floors from the beach during the nesting season is also recommended here. Mitigation measures should also be implemented for other Mediterranean nesting beaches affected by coastal development (Table 6.1). Such measures have already been successfully realized in the US, and should be an important part of European conservation efforts for sea turtles. The legal framework for these measures is provided by numerous European and national agreements protecting sea turtles, such as the Berne Convention and the Convention on Migratory Species.

1 General Introduction and State of Knowledge

Sea turtles (Cheloniidae, six species and Dermochelyidae, one species) are highly migratory reptiles with a wide range of distribution (summarized in RIEDE, 2001a, 2004), making the control of compliance with national and international conventions for conservation difficult. The IUCN (International Union for Conservation of Nature and Natural Resources) Red List of Threatened Species listed six of seven species in categories "Endangered" and "Critically Endangered" in 2006. For *Natator depressus* data was deficient, due to a lack of historical information (IUCN, 2006). All species are also listed in CMS (Convention on Migratory Species) Appendices.

Table 1.1: Sea Turtles: IUCN global status listing (2006)

Species:	Category[1]	Year Assessed
Cheloniidae		
Caretta caretta	**EN A1abd**	**1996** (IUCN version 2.3, 1994)
Chelonia mydas	**EN A2bd**	**2004** (IUCN version 3.1, 2001)
Eretmochelys imbricata	**CR A1bd**	**1996** (IUCN version 2.3, 1994)
Lepidochelys kempi	**CR A1ab**	**1996** (IUCN version 2.3, 1994)
Lepidochelys olivacea	**EN A1bd**	**1996** (IUCN version 2.3, 1994)
Natator depressus	**DD**	**1996** (IUCN version 2.3, 1994)
Dermochelyidae		
Dermochelys coriacea	**CR A1abd**	**2000** (IUCN version 2.3, 1994)

[1] **Criteria for Critically Endangered (CR)**:
A1: An observed, estimated, inferred or suspected reduction of at least 80% over the last three generations (IUCN Red List version 2.3, 1994)

Criteria for Endangered (EN):
A1: An observed, estimated, inferred or suspected reduction of at least 50% over the last three generations (IUCN Red List version 2.3, 1994)
A2: An observed, estimated, inferred or suspected reduction of at least 50% over the last three generations (IUCN Red List version 3.1, 2001)

Data Deficient (DD): Inadequate information to make a direct or indirect assessment of its risk of extinction, based on its distribution and/or population status (IUCN Red List version 2.3, 1994)

Addendum: in September 2007, the IUCN Standards and Petitions Working Group officially accepted the MTSG's (Marine Turtle Specialist Group) proposed listing of the Olive Ridley sea turtle, (*Lepidochelys olivacea*) as **Vulnerable (A2bd)**

Addendum: in April 2008, the IUCN Standards and Petitions Working Group officially accepted the MTSG Red List status assessment of the Hawksbill turtle (*Eretmochelys imbricata*) as **Critically Endangered (A2bd)**

This thesis focuses on sea turtle conservation at nesting sites. The primary objective is the investigation of light pollution, which is known to affect sea turtles and their hatchlings (WITHERINGTON and BJORNDAL, 1991b, WITHERINGTON, 1992a). A secondary objective is to test the possible effects of sounds on hatchlings on land. The basic knowledge of sea turtle sensory biology is incomplete, but better understood for *Chelonia mydas* and *Caretta caretta* compared with other sea turtle species (BARTOL and MUSICK, 2003). Thus, the species choice in my thesis allows a better evaluation and comparability of data obtained in the field and in behavioural experiments. Furthermore I investigated these two species because they represent the only sea turtles also nesting regularly in the Mediterranean, with larger populations found in Greece (MARGARITOULIS et al., 2003), Turkey, including N-Cyprus (CANBOLAT 2001, 2004), Cyprus (KASPAREK, 2001), and in Libya (LAURENT et al., 1999). My research should contribute to designing mitigation measures at those sea turtle nesting beaches affected by coastal development. As for Greece, Cyprus and Turkey, the latter being in European Union accession negotiations, focussing on these European-Mediterranean sea turtle populations was found to be valuable in order to provide a scientific approach for addressing conservation needs to the responsible authorities. Basic recommendations are drafted here (Chapter 6).

Compilation of sea turtle index nesting sites

"Index" (or "key") nesting sites are defined as major nesting areas of known status, or lesser nesting areas for which time-series quantitative data are available (SEMINOFF, 2004a). At the onset of this thesis in early 2005 few open access GIS (Global Information System) databases were available which could be used for creating *C. caretta* and *C. mydas* nesting site distribution maps. Therefore, compilation of *C. caretta* and *C. mydas* index nesting sites was the first step here. GROMS (Global Register of Migratory Species), a database for all migratory species as per definition (RIEDE, 2001a), provides general distribution maps for sea turtles but does not specify index nesting beaches. SWOT (State of the World's Sea Turtles), a project launched by a coalition of worldwide conservationists and scientists in 2003, drew attention to the lack of a joined-up database for sea turtles. First maps were available for *Dermochelys coriacea* in 2006. By now SWOT provides global nesting site distribution maps for four species, the Kemp's Ridley (*L. kempi*), which has a relatively small distribution in the Gulf of Mexico, the Loggerhead (*C. caretta*), The Hawksbill (*E. imbricata*) and the Leatherback (*D. coriacea*), providing information

about site location, nesting beach length, number of nests, methods of beach monitoring and the site contact person (SWOT, 2006, 2007, 2008). Distribution maps for *L. olivacea* and *C. mydas* are in preparation and not yet available. In my thesis I made a ranking of the ten largest nesting sites for *C. caretta* and *C. mydas*, as per nesting females/year and per country. These data were compiled by literature research and cross-checked for consistency with the latest data available, using the SWOT report for *C. caretta* (SWOT, 2007) and the IUCN Red List assessment for *C. mydas* (SEMINOFF, 2004a), the latter indicating 32 index nesting sites. Collated data for the two species was used to create global nesting site distribution maps in Arc View GIS for comparison with special light pollution maps (Figs. 3.3 – 3.7). With a focus on European species conservation, detailed nesting site distribution maps were provided for *C. caretta* and *C. mydas* index sites in the Eastern Mediterranean, highlighting light pollution hotspots (Figs. 3.9 – 3.13, 3.15, 3.16).

Coastal Development: A major threat to sea turtles

Sea turtle females are known to periodically return to the same beaches for nesting, generally the beaches they left as hatchlings (MROSOVSKY, 1983, MILLER, 1997). These nesting habitats are critical for reproduction and survival of the species. Consequently, this high nesting-site fidelity makes nesting beaches the weak points in sea turtle life cycles. Though some local sea turtle populations show recovery due to intensive conservation efforts, global populations are considered to be declining (IUCN, 2006). The Marine Turtle Specialist Group (MTSG), conducting global status assessments of sea turtle species to be included in the IUCN Red List, lists five major threats to sea turtles[1]:

1. Fisheries impacts
2. Direct take
3. Coastal Development
4. Pollution and pathogens
5. Global warming

These threats will result in further decline, local extinction, and/or will prevent recovery of sea turtles if uncontrolled.

Coastal development, as a concomitant of increasing urbanization and mass tourism, includes beach armouring (flanking of beaches with concrete or rocks, e.g. to protect

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[1] www.iucn-mtsg.org/

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from erosion), beach nourishment (artificial replacement of soil or sand lost due to erosion) and sand mining (removal of sand for e.g. construction activities). Coastal infrastructure like construction sites, hotels and other buildings, watersport areas and roads are also known to have negative effects on the ecosystem and sea turtle nesting populations in the Mediterranean (MARGARITOULIS et al., 2003, CORBETT and KASPAREK, 2003, VENIZELOS, 2001). A case study is presented in Chapter 4.

Sea turtles and light pollution: A key problem at nesting beaches

Sea turtle hatchlings orient towards natural light cues at the broad open horizon of the sea, where star- and moonlight is reflected at night, and away from elevated shapes like dunes and vegetation. This natural light is known to be the primary orientation cue for hatchlings, guiding them to the sea. The mechanism of visual orientation in hatchlings is summarized in SALMON and WYNEKEN (1994):

1. After emerging from underground nests, sea turtle hatchlings generally orient towards the brightest direction.

2. Sea turtle hatchlings move away from high silhouettes, e.g. dunes and beach vegetation.

3. When visual cues are indistinct, sea turtles move in relation to elevation, not brightness.
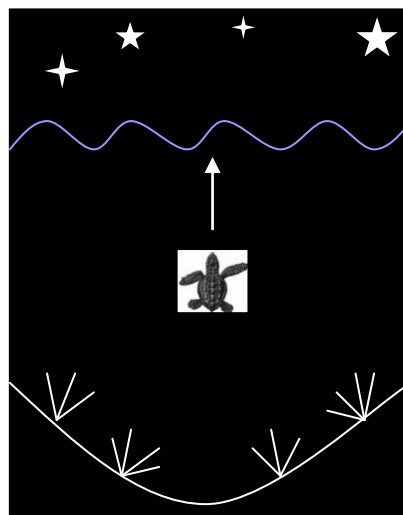
Figure 1.1: Hatchling seaward orientation at night, under natural light conditions.

A key problem at sea turtle nesting beaches is light pollution, the illumination of the beach and the night sky by artificial lights. Artificial lights set landwards may attract hatchlings on their crawls to the ocean, by providing more intensive stimuli and eliciting positive phototaxis. This disorientation may result in delayed sea finding or

even death due to exhaustion, dehydration or predation (MCFARLANE, 1963, PHILIBOSIAN, 1976). Consequently, the chance of hatchling survival at beaches with intense artificial lighting is reduced (SALMON, 2003, WITHERINGTON, 1997). Artificial lights may also disturb female sea turtles emerging from the sea for nesting (WITHERINGTON, 1992a).

For an understanding of the visual capabilities of sea turtles and their hatchlings, a number of studies on morphology, electrophysiology and behaviour have been conducted in the past years (summarized in BARTOL and MUSICK, 2003). These data confirm that sea turtles have evolved a complex vision system that is able to perceive a wide spectrum of light. LIEBMAN and GRANDA (1971) compared light absorption in visual pigments in a freshwater species, the Red Eared Slider (*Pseudemys scripta elegans*), and a sea turtle, the Green Turtle (*Chelonia mydas*). They found that both species have a duplex retina containing rods and cones. Three photo pigments were found associated with photoreceptor cells, in *C. mydas* absorbing light maximally at 440 nm, 502 nm and 562 nm. The rods were found to absorb light maximally at 502 nm. In contrast, rod and cone sensitivity in the freshwater turtle was shifted towards the higher visible wavelengths at 450 nm (cones), 518 nm (rods and cones) and 620 nm (cones). The authors concluded that this discrepancy might be an adaptation to the different habitats used by the two species, as seawater transmits shorter visible wavelengths at greater depths compared with freshwater. GRANDA and O`SHEA (1972) measured the electrical responses of photoreceptor cells in dark-adapted turtle eyes. They found that *C. mydas* had highest sensitivity at 450-450 nm, 520 nm and 600 nm. Discrepancies in spectral sensitivity, compared with data from LIEBMAN and GRANDA (1971), were attributed to coloured oil droplets in the cones of *C. mydas*, yellow ones in the 502 nm photo pigment and orange ones in the 562 nm photo pigments. Clear oil droplets were only found in the 440 nm photo pigment, which did not cause a shift in the absorbed spectral sensitivity. A possible function of oil droplets is that of natural cut-off filters, bundling light of specific wavelengths for better processing in photoreceptors (GRANDA and O`SHEA, 1972). Behavioural studies conducted on *C. caretta* and *C. mydas* hatchlings support increased vision at 400 nm and 500 nm and also in the near-ultraviolet range at 360 nm (WITHERINGTON and BJORNDAL, 1991a). A recent study on *C. mydas* confirms that the retina of sea turtles includes a UV-receptor, making them tetrachromats (MÄTHGER et al., 2007).

Table 1.2: Spectral sensitivity of sea turtles

Species (life stage)	Spectral sensitivity[1]	Method used	Author
C. mydas (adults)	440 nm, 502 nm, 562 nm	Microspectrophotometry	LIEBMAN and GRANDA (1971)
C. mydas (adults)	450-460 nm, 520 nm, 600 nm	Electroretinography	GRANDA and O`SHEA (1972)
C. caretta, C. mydas (adults)	400-700 nm (*C. mydas*) 440-700 nm (*C. caretta*) both species had peaks at 580 nm	Electroretinography	LEVENSON et al. (2004)

[1] Discrepancies of wavelength measurements in the two older studies were attributed to the interaction of visual pigments and the cone oil droplets, which have light-filter function (GRANDA and O`SHEA, 1972).

Species (life stage)	Sensitivity range[2]	Method used	Author
C. caretta (hatchlings)	360 nm to 700 nm	Behavioural study (orientation experiment in two-choice box)	WITHERINGTON and BJORNDAL (1991a)
C. mydas (hatchlings)	360 nm to (at least) 500 nm	Behavioural study (orientation experiment in two-choice box)	WITHERINGTON and BJORNDAL (1991a)

[2] *C. mydas* responded insignificantly to 600 nm (yellow-orange) and 700 nm (red), *C. caretta* showed aversion to 560-600 nm (yellow), assessed as xanthophobia (WITHERINGTON and BJORNDAL, 1991a).

Increased perception of shorter visible wavelengths < 600 nm is the key for understanding sea turtle hatchling behaviour at nesting beaches. The problem of hatchling disorientation caused by artificial lights was investigated in detail in experiments from the 1960s onwards (MROSOVSKY and CARR, 1967, MROSOVSKY and SHETTLEWORTH, 1968). Investigation in the field revealed that in fact most light sources used in outdoor lighting at *C. caretta* nesting sites in the United States emitted in the short visible wavelength spectrum, which was attracting sea turtle hatchlings. Data on disruptive light sources are summarized in Table 1.3.

Table 1.3: Polychromatic light sources affecting sea turtles

Model	Wavelengths emitted (peaks)	Used for
Metal Halide	550 to 625 nm	Outdoor lighting
Mercury Vapour	554 to 575 nm	Outdoor and street lighting
High Pressure Sodium Vapour (HPS)	500, 570 to 630 nm	Street lighting

For conservation measures, *C. caretta* index nesting beaches in the United States underwent intense investigation of light pollution. Precise proposals for mitigation measures were made from the late 1990s onwards, including replacement of polychromatic High Pressure Sodium Vapour (HPS) by Low Pressure Sodium Vapour (LPS) lights. These have more monochromatic properties at 590 nm and were found to be less attractive to hatchlings. A detailed compilation of technical data is provided in WITHERINGTON and MARTIN (1996) and in the technical report of the Florida Power and Light Company (2002).

In contrast to the United States there are only a few studies available investigating the effects of light pollution at nesting beaches in the Mediterranean (PETERS and VERHOEVEN, 1994, IRWIN et al., 1996). Also proposals for effective reduction of light pollution at the Mediterranean sea turtle index nesting beaches have not yet been considered (DEMETROPOULOS, 2001, VENIZELOS, CANBOLAT, pers. comm.). Therefore the primary objective of my thesis is the overall assessment of light pollution and its interactions with sea turtles and their hatchlings in this important nesting region. It is suggested that increasing mass tourism will accelerate coastal development in Greece, Cyprus and Turkey, which hold the majority of *C. caretta* and *C. mydas* index nesting beaches in the Mediterranean (VENIZELOS, 2001). As a consequence, light pollution will also increase on the Mediterranean coastline. In fact, using satellite data on light pollution from CINZANO et al. (2001a), I was able to identify a high number of light pollution hot spots at *C. caretta* and *C. mydas* index nesting beaches in this region. Considering these trends, research efforts on the impact of light pollution on sea turtles are important for the future conservation of these species here. In a case study I made a census of stationary artificial light sources that were contributing to high light pollution levels in Belek, Turkey. This is one of the most important *C. caretta* sites in the Mediterranean. In addition, I made an evaluation of the impact of polychromatic lights on the nest site selection of females, hatchling

orientation and hatchling mortality at this index nesting site. The goal was to recommend reasonable and practical mitigation measures for improved sea turtle conservation at Belek. My research should also contribute to initiating further investigation on light pollution and its interactions with sea turtles in the Mediterranean.

Sea turtles and noise pollution: A possible problem at nesting beaches?

Hearing in sea turtles has not been studied extensively, but indications are that the turtle ear is relatively insensitive to high frequencies and functions best below 1 kHz. Sea turtle ear morphology and physiology is described in several publications (WEVER and VERNON, 1956, TURNER, 1978, WEVER, 1978, LENHARDT, 1982). LENHARDT et al. (1983) measured neural responses of *Caretta caretta* and *Lepidochelys kempi* to underwater sound and concluded, that "bone conducted hearing" appears to be a reception mechanism for marine turtles, with the skull and shell acting as receiving surfaces. Another study on tortoises indicated that neural electrical responses of Box Turtles (*Terrapene carolina*) to vibrations were lower in threshold and higher in amplitude than responses to air-conducted clicks, suggesting that the turtle ear is differently sensitive to aerial sound and vibration (LENHARDT and HARKINS, 1983). Dissections and examinations of the middle ear structures of five adult *C. caretta* and one *L. kempi* confirmed that the thick tympanum (Figure 1.3), while disadvantageous as an aerial receptor, likely enhances low-frequency bone-conducted hearing (LENHARDT et al., 1985). Newer preliminary data on sea turtle inner ear anatomy indicate that there may be a shift in the mechanism of sound perception between different life stages (LENHARDT, 2005). The same author suggests that adults rely mainly on bone-conducted hearing, whereas hatchlings have increased hearing in air, which is supported by the higher density of basilar membrane hair cells in the hatchling's inner ear. Electrophysiological studies conducted on sea turtles confirm that they have a low-frequency receptor ear, which is able to perceive both air-conducted sounds and vibratory stimuli (RIDGWAY et al., 1969, BARTOL et al., 1999, BARTOL and KETTEN, 2006). RIDGWAY et al. (1969) measured the auditory sensitivities of juvenile *C. mydas*, which is the only investigation using aerial sound so far (Table 1.4, Figure 1.2).

Table 1.4: Hearing ranges of sea turtles (electrophysiological data)

Species (life stage)	Hearing range	Stimuli used	Author/year
C. mydas (juveniles)	60 -1000 Hz	Air-conducted sounds and vibrations[1]	RIDGWAY et al. (1969)

[1] RIDGWAY et al. (1969) measured cochlear potentials elicited by pure tones and vibratory stimuli up to 2000 Hz. They found maximum sensitivity to be from 300-400 Hz (Figure 1.2).

Species (life stage)	Hearing range	Stimuli used	Author/year
C. caretta (juveniles)	250 – 750 Hz	Vibrations[2]	BARTOL et al. (1999)

[2] BARTOL et al. (1999) used the non-invasive technique of ABR (Auditory Brainstem Responses). Two different vibratory stimuli were used, broadband low frequency clicks and pure tone bursts. Maximum hearing sensitivity was measured at 250 Hz.

Species (life stage)	Hearing range	Stimuli used	Author/year
C. mydas (subadults)	100 - 500 Hz	Water-conducted sounds[3]	BARTOL and KETTEN (2006)
C. mydas (juveniles)	100 – 800 Hz	Water-conducted sounds	BARTOL and KETTEN (2006)
L. kempi (juveniles)	100 – 500 Hz	water conducted sounds	BARTOL and KETTEN (2006)

[3] BARTOL and KETTEN (2006) used ABRs on sea turtles, which had their ears submerged underwater. Pure tones were used delivered by a sound source located above the animal. Maximum hearing sensitivity was found at 300 Hz (subadult *C. mydas*) and 600 Hz (juvenile *C. mydas* and *L. kempi*).

FIG. 2.—Sensitivity curves for aerial sounds, obtained in two specimens of *Chelonia mydas*. The sound pressure is shown, in decibels relative to 1 dyne/cm,² required to produce a cochlear potential of 0.1 μv.
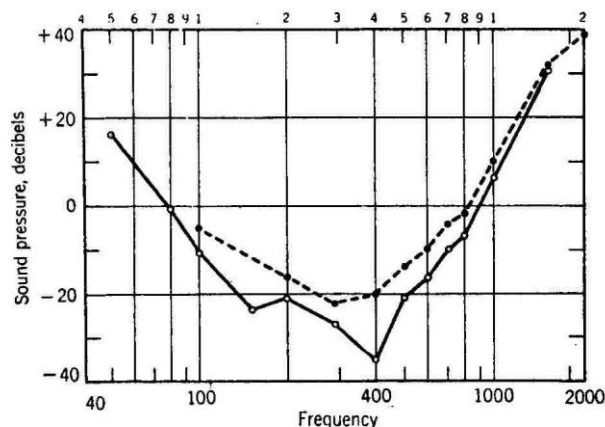
Figure 1.2: Juvenile *C. mydas* hearing sensitivity curves for aerial sounds (from Ridgway et al., 1969). Maximum sensitivity was measured at 400 Hz.
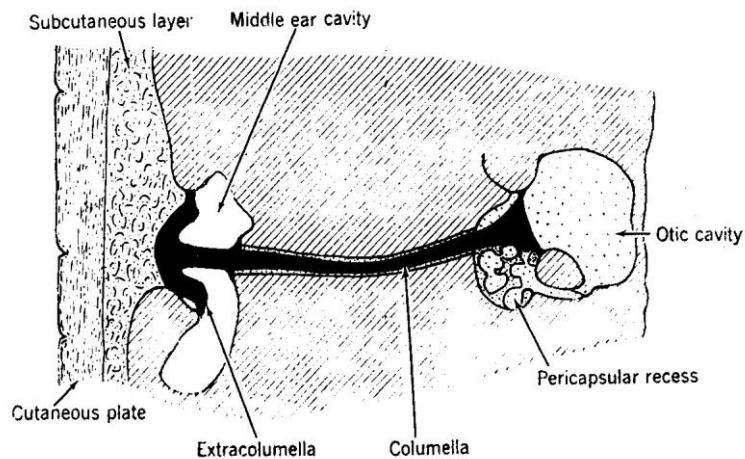
FIG. 1.—The auditory mechanism in *Chelonia mydas*. The left ear is shown in a dorsal view. Scale: ×2.

Figure 1.3: Structure of the juvenile *C. mydas* ear (from Ridgway et al., 1969). While lacking an external ear, the middle and inner ear structures are clearly developed. The cutaneous plate serves as tympanum, separated from the two ossicular structures (Extracolumella and Columella) by a thick subcutaneous fat layer. The columella is connected to the cochlea, which holds a fluid in the otic cavity for pressure relief.

There is a general lack of auditory data for sea turtles at different life-history stages. Until now, hearing ranges have been investigated only in sea turtle juveniles, which are the post-hatchling life stage, and in a limited number of subadults. For hatchlings there is no data on hearing ranges available as yet. Altogether very little is known about basic hearing mechanisms or the role of sound in sea turtle life cycles. The turtle brain centre, which serves for processing acoustic signals, is relatively small, and may not enable complex functions. In fact acoustic communication probably is not common in turtles. Few species are known to produce sounds, which are believed to be mainly incidental during mating or nesting (GANS and MADERSON, 1973) or when disturbed (CAMPBELL and EVANS, 1972). Leatherback (*D. coriacea*) females are known to produce low intensity sounds when nesting, with peaks between 300 and 500 Hz (COOK and FOREST, 2005). The authors do not exclude that these sounds might have communicatory functions, because the sounds generated coincide with the spectrum in which sea turtles perceive sounds (RIDGWAY et al., 1969, BARTOL et al., 1999). It is unclear what function acoustic perception in sea turtles on land may have and as yet no studies have been done on this matter. Sea turtles may also be capable of perceiving the low-frequency spectrum of the natal beach, which could even serve as one of the cues in nesting returns (LENHARDT et al., 1983). This hypothesis is supported by preliminary findings of NUNNY et al. (2005). The latter authors suggest that females use wave sounds to control their nesting behaviour. If this is shown to be the case it remains unclear if sea turtles are

imprinted to these natural sounds as hatchlings already in the nest, assuming that they can perceive substrate vibrations from the surf there. Other mechanisms would involve perception of vibrations or aerial sounds after emerging, while crawling on land, or detection of underwater sounds when swimming offshore. In this context the possible role of natural wave/surf sounds as orientation cues for hatchlings heading seawards is also speculative. Preliminary data on *Caretta caretta* hatchling acoustic orientation indicate that hatchlings do not orient to experimental aerial wave sounds (MANGIAMELE and LOHMANN, 2005). But this is the only study done so far and these data need verification (see Chapter 5).

A concomitant of coastal development is anthropogenic noise, known as noise pollution. Noise pollution affects animals in many ways, ranging from annoyance and chronic stress to hearing loss (BOWLES, 1995, MOEIN et al., 1995). Whereas there is evidence that noise has a negative impact on submerged sea turtles (O'HARA and WILCOX, 1990, MOEIN et al., 1995, MCCAULEY et al., 2000), there is a lack of scientific data on the effects which noise has on sea turtles while on land, justifying basic research on this issue. Two behavioural studies on juvenile *C. caretta* show that avoidance responses are elicited by low frequency underwater sounds at high intensities (O'HARA and WILCOX, 1990, MOEIN et al., 1995). A newer study, measuring underwater low frequency sounds in a sea turtle foraging habitat, confirms that human activity increases the underwater ambient noise considerably. The authors suggest that noise pollution may affect sea turtles (SAMUEL et al., 2005). Further adverse effects of noise on turtles are mentioned in the literature, ranging from behavioural modification (including mild disturbance, disruption or impairment of activities, and displacement from key habitats), to injury, disorientation, capillary damage, loss of motor control, and even to death in severe cases (LENHARDT, 1994; LUTCAVAGE et al., 1997). A study conducted by the Greek National Aviation Service (1990) indicates that high-intensity aircraft sounds at the *C. caretta* index nesting site at Laganas Bay on Zakynthos, Greece, have a negative impact by deterring nesting females, but these data are not available in English (Medasset, pers. comm.).

A first step in my research on auditory perception was to find out if hatchlings in fact are able to perceive low frequency surf sounds, which is the predominant sound source at coastal habitats (WILSON, 1998, LOEWEN and FARELL, 1998). Also it was tested in experiment if hatchlings show behavioural modification in the presence of acoustic stimuli. My research approaches and results are outlined in Table 1.5.

Table 1.5: Research approach to test a possible impact of sounds on sea turtle hatchlings

Literature data available to date	Theoretic approach (research objectives)	Experimental approach in this thesis	Technical approach in this thesis
Electrophysiological data on sea turtle acoustic perception confirm sensitivity for low frequency sounds up to 1 kHz. Mechanisms: **Air conduction** (RIDGWAY et al., 1969) **Bone conduction** (vibratory stimuli) (RIDGWAY et al., 1969, BARTOL et al., 1999) **Water/bone conduction** (BARTOL AND KETTEN, 2006)	Low frequency sound perception on land is in principle possible (air conduction, vibrations). Surf/wave sound perception is also possible.		A) Measurements on surf sounds and low frequency noise at a *C. caretta* nesting beach (Table 4.15) B) Comparison of surf sounds measured with electrophysiological data for *C. mydas* (RIDGWAY et al., 1969) (Tables 4.5, 4.6). Comparison confirms that surf sounds in 20 m distance are within the hearing range of *C. mydas* juveniles
Behavioural data: *C. caretta* hatchlings do not orient towards aerial surf sounds (MANGIAMELE and LOHMANN, 2005). **Morphological** data: Hatchlings have fully-developed basilar hair cells, supporting increased perception of aerial sounds compared with later life stages (LENHARDT, 2005).	Like *C. mydas* juveniles, sea turtle hatchlings may be capable of perceiving low frequency sound on land (i.e. surf/wave sound).	Field experiments conducted with *C. caretta* hatchlings. Test results with small sample size revealed that hatchlings significantly failed to orient seawards in presence of artificial lights and sounds, when shielded against substrate vibrations from the shore.	Wooden experimental arena structure with shielding properties against artificial light and substrate vibrations (Fig. 4.20). Tests conducted in Belek, Turkey
To date, no behavioural test conducted on hatchlings has been made using aerial wave/surf sounds <u>and</u> vibratory stimuli simultaneously.	Hatchlings may use acoustic cues, aerial sounds and/or substrate vibrations for **orienting seawards**.	*C. caretta* and *C. mydas* hatchling behaviour was tested in the presence of aerial and vibratory sound stimuli under experimental conditions.	Wooden two-choice box with low frequency speaker using **wave/surf sound** recordings from disc (Fig. 5.5). Tests done in Xcacel, Mexico
To date, no behavioural test has been conducted on hatchlings investigating a possible **negative impact of noise** in experiment.	Noise pollution may have a negative impact by disorienting hatchlings (masking of surf sounds) or causing other behavioural modification.	As above. Neither species showed significant aversion or attraction to sound stimuli, whereas there was evidence for inhibition of motor activity.	As above, but experimental **noise** used (Pink noise, street traffic sounds, fireworks noise, 400 Hz and 1000 Hz test tones)

Rationale for investigating sea turtle hatchlings' wave/surf sound perception ability:

Hatchlings may be able to detect low frequency sounds of the seashore and use them as secondary orientation cues when visual cues are indistinct, in given scenarios:

A) Hatchlings emerging in the late evening or in the early morning exposed to natural daylight. Exclusive visual orientation may lead to disorientation if orienting towards the sun shining from landwards at a low visible angle (Figure 1.4).

B) Hatchlings emerging during the night in the presence of a bright moon shining from landwards. Visual orientation to the brightest direction or away from shadows may lead to disorientation (Figure 1.5).

C) Hatchlings emerging when visual cues are weak or absent, e.g. in the presence of diffuse light during overcast weather conditions or nearly dark sky at new moon phases. Hatchlings may not be able to orient visually at all (Figure 1.6).

D) Hatchlings emerging behind beach vegetation or dunes, preventing a view of the open ocean. Exclusive visual orientation may not be possible (Figure 1.7).
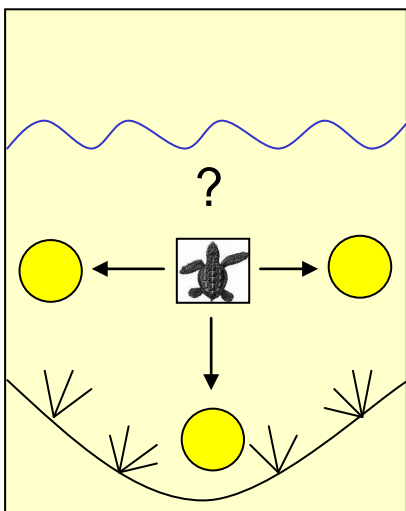
Figure 1.4: Scenario A
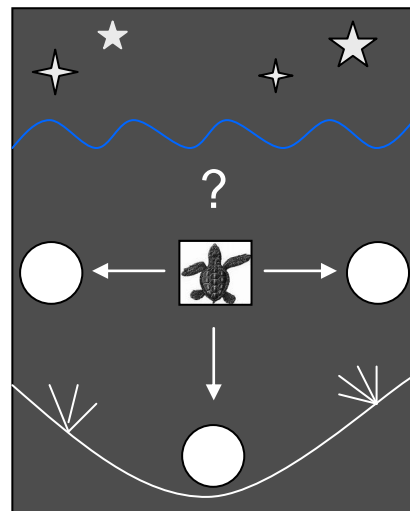
Figure 1.5: Scenario B
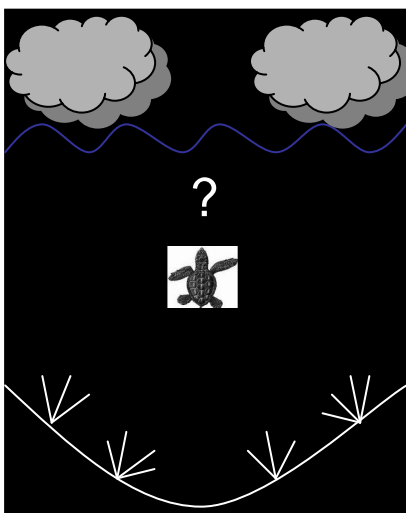
Figure 1.6: Scenario C
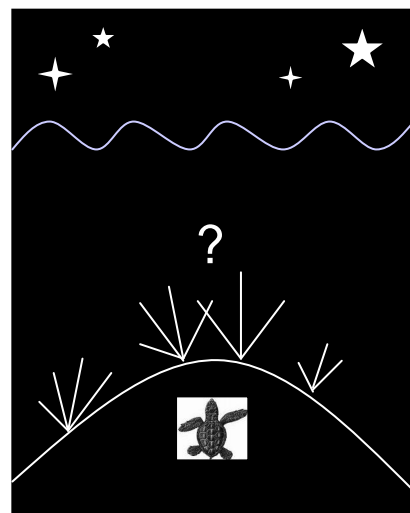
Figure 1.7: Scenario D

Scenario A: Hatchling orientation towards the sun is controversial. MROSOVSKY (1970) found that the sun attracted *Chelonia mydas* hatchlings. In Belek, Turkey, I observed that *Caretta caretta* hatchlings, which were released on the beach in the morning, oriented towards the sun visible from the east (landwards) at an estimated vertical 30º angle. VAN RHIJN (1979) for *C. mydas* and WITHERINGTON (1992b) for *Caretta caretta* made contradictory findings. These authors concluded that the sun did not affect hatchlings.

Scenario B: As for the sun, hatchling orientation towards a bright moon visible at a low vertical angle may be possible, whereas the moon at its zenith probably may not be detectable, considering the relatively narrow vertical cone in which hatchlings perceive light. This is believed to be "a few degrees" for *Chelonia mydas* and *Lepidochelys olivacea* (VERHEIJEN and WILDSCHUT, 1973) and between 10º and 30º for *C. caretta* (SALMON and WYNEKEN, 1994, WITHERINGTON, 1992b). Besides having a possible impact by attracting hatchlings, when the moon is close to the horizon and visible to hatchlings (WITHERINGTON, 1992b) increased ambient moonlight plays a major role in reducing hatchling disorientation in developed beach areas, since it is mitigating the adverse effects of light pollution. This is shown in my thesis at a *C. caretta* nesting beach affected by light pollution (Chapter 4.1.3).

Scenario C: Weather conditions, such as a change in cloud cover, are known to affect the straight seaward orientation of *C. mydas* hatchlings (MROSOVSKY and SHETTLEWORTH, 1968).

Scenario D: Though most sea turtles nest close to the shore in front of dunes and beach vegetation, some species also nest further landwards. This is known for *Dermochelys coriacea,* but is more typical for *Eretmochelys imbricata*. Sea-finding tests for these two species showed that hatchlings were more susceptible to disorientation in vegetated areas, or areas deeper in the forest (KAMEL and MROSOVSKY, 2004, 2005).

Whether hatchlings have specially developed mechanisms (e.g. acoustic orientation) to compensate for impeded sea-finding, or generally exhibit lower sea-finding in the presence of natural brightness cues from landwards (Scenarios A, B), or in absence of clear visual cues (Scenarios C, D), has not been thoroughly investigated and needs verification (see Table 1.5).

Rationale for investigating the effect of noise pollution on hatchlings:

Under the presumption that sea turtle hatchlings are capable of perceiving wave/surf sounds and use them as orientation cues, anthropogenic low-frequency noise may have a negative impact on hatchling orientation, e.g. by masking wave/surf sounds.

Acoustic noise is defined as sound from an anthropogenic source. The scale used for measuring noise is the sound pressure level (SPL) in decibel (dB), comparing the ambient pressure in a medium, e.g. air, with the standard reference pressure. For air this is 0 dB (SPL), equivalent to 20 µ Pascal, a value defined as the threshold level of human sound perception. In contrast, the sensitivity for air-conducted sounds in freshwater turtles, *Pseudemys scripta*, is 30 to 40 dB (SPL) lower (PATTERSON, 1966). This means turtles generally need higher sound levels for perception of aerial sounds compared with humans. Sea turtles' average sensitivity for airborne sounds is probably only > 50 dB (SPL) (RIDGWAY

et al. 1969, LENHARDT, 1994). Thereafter it is unclear if wave sounds and anthropogenic noise at a beach can generally be perceived by emerging sea turtle hatchlings. For verification I measured ambient sounds and noise pollution in the field and made comparisons with electrophysiological data available for juvenile *C. mydas* (RIDGWAY et al., 1969). This data was used since comparable electrophysiological data for sea turtle hatchlings is not available to date.

For investigating the impact of noise on sea turtles, the knowledge of the sound source is crucial for making an assessment of its effect over a distance. For point sources, like discotheque noise, the SPL attenuates by 6 dB per doubling of distance. In contrast, the SPL of a line source, such as natural surf sound, attenuates only by 3 dB per doubling of distance. Thus, at the same SPL and distance to a sea turtle nest, a point source (noise) would be less likely to affect the audibility of the surf (masking effect), since it attenuates much faster over the distance. However, a point source at high SPL in close proximity to a sea turtle nest may mask surf sound and make it unidentifiable for hatchlings. This is under the precondition that it has a similar frequency range (PATTERSON and GREEN, 1978).

In Belek, Turkey, I measured SPL and frequency curves of surf sounds, discotheques and street traffic noise. I found that the SPL of some noise sources were higher compared with the surf sound, if located close to sea turtle nests (Figure 1.8). Frequency spectra of discotheque noise had distinct peaks in the low frequency range < 1000 Hz, similar to surf sounds (Chapter 4.1.3). Therefore, under the precondition that hatchlings are able to perceive and orient towards surf sounds, low frequency noise at high intensities might be a risk to hatchlings. It may cause disorientation by masking natural acoustic cues or other behavioural modifications such as avoidance or startle responses, as observed for submerged sea turtles (LENHARDT, 1994; LUTCAVAGE et al., 1997, SAMUEL et al., 2005).

Adult sea turtle females are known to avoid nesting beaches with a high degree of noise pollution. Whether this is due to individual aversion or a congenital protective function for hatchlings is highly speculative. Females, while not providing brood care, may instinctively avoid noisy beaches, just as they avoid beaches with high light pollution levels (WITHERINGTON, 1992a), to protect their offspring.
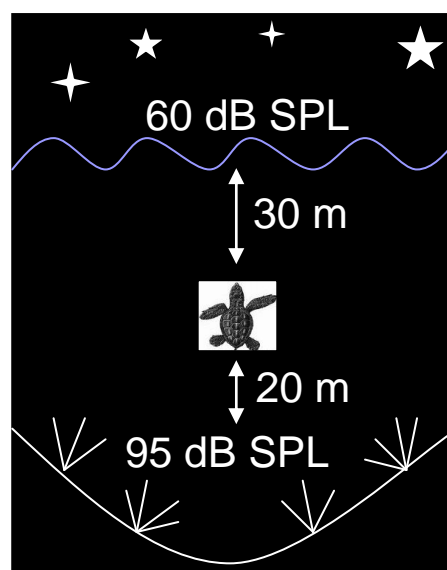
Figure 1.8: Scenario E: Discotheque noise at close proximity to a sea turtle nest, masking surf sounds

2 Compilation of global sea turtle nesting sites with a focus on the Mediterranean

2.0 Introduction

Status assessments for sea turtle populations are difficult (SEMINOFF, 2004b, MAST et al., 2006, NARO-MACIEL and FORMIA, 2006). This was evidenced by outdated IUCN Red List data for five of seven sea turtle species, and data deficiency for one species (IUCN, 2006, see Table 1.1). Due to this general lack of information, when this work commenced there was no database available that summarized the current status of the nesting populations of all species globally. In recent years research efforts have been made to compensate for this lack of knowledge. The Marine Turtle Specialist Group (MTSG) presented a Top ten list of the most threatened sea turtle populations on a regional level, including *C. mydas* in the Mediterranean (MAST et al., 2005). However, sea turtle distribution maps providing detailed and up-to-date nesting population data are sparse. The UNEP-WCMC Marine Turtle Interactive Mapping System[2] provided nesting data and migration routes for six sea turtle species only for the Indian Ocean and the Mediterranean. But these data are partly incomplete and outdated, and there is no data available for the Atlantic and Pacific Ocean sea turtle populations. Realizing the need for a centralized database, the SWOT (State of the Worlds Sea Turtles) project was launched in 2003. This was an important step in summarizing information from sea turtle scientists and conservationists on a global scale, using GIS (Global Information System) data to develop nesting distribution maps. First reports compiled detailed nesting data for *Dermochelys coriacea* in 2004 (SWOT, 2006), *Caretta caretta* in 2005 (SWOT, 2007), and *Eretmochelys imbricata* in 2006 (SWOT, 2008). On the SWOT website[3], index beaches are also summarized for *Lepidochelys kempi*, which has a restricted nesting distribution range within the Gulf of Mexico. Nesting distribution maps are still lacking for *Chelonia mydas*, *Lepidochelys olivacea* and *Natator depressus* in the SWOT database. My thesis does not go beyond its scope to list all sea turtle nesting sites known from literature. It compiles current nesting site distribution data for *Caretta caretta* and *Chelonia mydas*, which are the species discussed here. A focus is set on the Mediterranean, which is of global importance for *C. caretta*, but also for a small local *C. mydas*

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[2] http://stort.unep-wcmc.org/imaps/indturtles/viewer.htm

[3] http://www.SeaTurtleStatus.org

subpopulation, which was categorized as "critically endangered" in this region (IUCN, 2004). In combination with light pollution maps for the Mediterranean (Chapter 3), compilation of index nesting sites is understood as a research tool. It is a first step to identify light pollution at nesting sites on a global scale in order to highlight conservation needs and to initiate further investigation into its interaction with sea turtles.

Sea turtle life cycle and nesting beach monitoring

Sea turtle life cycles were only incompletely known for a long time and gaps in our knowledge still exist. After hatching from underground nests, sea turtle hatchlings head to the sea in a state of increased activity, known as the "frenzy" phase (WYNEKEN and SALMON, 1992), then further offshore to their feeding and foraging grounds. Here they spend a pelagic phase of 5 to 20 years, which is known as the "lost years" (CARR, 1952). During this time hatchlings and small juveniles (post-hatchling life stage) are only sporadically detectable in the open sea. Juvenile sea turtles return to littoral waters for feeding. After reaching maturity, which may vary within species from 11 to 35 years in *L. olivacea* to 25 to 50 years in *C. mydas* (HIRTH, 1997), males and females migrate to their mating grounds. The females emerge at sandy beaches for nesting, while males keep waiting offshore for receptive females, or return to their feeding grounds. Females stay near the nesting beaches during the whole nesting season, usually several months, to lay several clutches. The number of clutches laid is variable within species, but usually between 2 and 4 per nesting season. For *C. caretta* this is an average of 4 nests (DODD, 1988), for *C. mydas* 3 nests per season (MILLER, 1997). After an incubation time of 50-70 days, which is dependent on temperature, hatchlings emerge from the nests within 48 hours, in one or more emergence events, usually at night. A new life cycle begins. Since nesting beaches provide adequate conditions to capture two life-history stages of sea turtles, hatchlings and the adult females, initial research efforts concentrated on obtaining census data on land. In the 1950s researchers started to count sea turtle tracks, number of females, eggs and hatchlings at nesting beaches and also assessed the morphometry of females. These methods are generally summarized under the term "nesting beach monitoring". Monitoring methods are variable worldwide and will be described in detail for the *C. caretta* nesting beach at Belek, Turkey, which I investigated in the field (Chapter 4). For identifying individual females, different marking techniques were developed over the years. A first step

was using "tags" on individuals' fore-flippers, hence the method is named "tagging". These tags, made of metal or plastic, hold the individual's identification number, the date of tagging, and name and address of the person or institution who tagged the individual. Using this method of "mark and recapture" over a long time, individuals could be clearly identified when returning to their beaches. Thus tagging females was the crucial step to assessing population sizes based on census data on beaches. Moreover it could be proven that females returned to the same nesting beaches over many years, a characteristic of sea turtles known as nesting site fidelity (MROSOVSKY, 1983, BOWEN et al., 1992, MILLER, 1997, BRODERICK et al., 2007). But many tags are lost during sea turtles' migrations, biasing population data since already identified females may be assessed as new individuals. Therefore, at nesting beaches newcomer females are controlled for cues of tag loss, such as typical scars on their fore-flippers. It is pointed out that methods of tagging are not yet standardized globally, which may also result in inconsistency when identifying individual females. Detailed information on the tagging methods used to date is provided on the seaturtle.org website[4]. Though tagging allows the obtaining of long-term data on nesting populations, it provides less information on sea turtles at sea. Researchers generally have limited access to migrating sea turtles, often relying on bycatch data given by the crews of fishing vessels. But bycatch data obtained by fishing activities is considered unreliable for representing actual population sizes, as it is often incidental and dependent on the fishing area, the method of fishery used, and also the species and its life stages. Nevertheless, the fishing industry, including flounder fishing, gill netting, crab trawling, longline fishery, driftnet fishing, pure seines and bottom trawl fishery, significantly endangers sea turtles in their benthic and pelagic life stages, both in North American and European waters (TEWG, 1998 and 2000; NMFS, 1998, 2001, WATSON et al., 2003). Incidental catch in shrimp fishery was the main human-caused mortality factor in Western North Atlantic *L. kempi* and *C. caretta* populations prior to the implementation of Turtle Excluding Devices (TED) in 1989, killing more benthic immature and adult sea turtles than all other human sources combined. There is also data available from longline fisheries, mainly concerning the carnivorous species *C. caretta* or *D. coriacea*, which are attracted by the baits (SHOOP & RUCKDESCHEL, 1982, AGUILAR, 1995). According to the European Commission Project 98/008 (LAURENT et al., 2001), fishery activity is also considered

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[4] www.seaturtle.org/tagging/

the most important anthropogenic mortality factor known for the endangered *C. caretta* and *C. mydas* breeding in the Mediterranean. European longline fisheries, primarily targeting tuna and swordfish, is known to trap up to 20 000 *C. caretta* per year in Spanish waters alone, killing approximately 10 700 of this species annually (AGUILAR, 1995). A further important step in understanding sea turtle population biology was the introduction of satellite telemetry on sea turtles from the late 1970s (STONEBURNER, 1982). Though not used to a large extent compared to tagging because of cost factors, this method allows the understanding of individual migration routes, and consequently the degree of genetic exchange that might occur between populations. Based on this method it was proven that sea turtles migrate thousands of kilometres between their feeding and nesting grounds, exhibiting an accuracy and continuity unique among reptiles (LUSCHI et al., 1996, PAPI et al., 1997). The methods of sea turtle satellite telemetry are not specified in detail here, since they are unrelated to methods used in my thesis. Further information is found on the seaturtle.org website and its STAT-Project (Satellite Tracking and Analysis Tool), which was founded in 2003 and enables data entry, evaluation, storage and file sharing from the ARGOS-satellite network. As for other species, mitochondrial (mt) and nuclear (n) DNA analysis is a tool also used for sea turtles, to assess the genetic origin of an individual. By taking blood or tissue samples for laboratory analysis, this method can be easily applied to all life history stages, on the beach and at sea. Comparison with the DNA profile of different populations has helped in assessing the importance of local populations for considering conservation measures, such as for the genetically distinct Mexican *C. caretta* and *C. mydas* populations at Xcacel (ENCALADA et al., 1999) which I investigated in the field in 2006 and 2007 (Chapter 5).

Assessment of sea turtle population sizes

For estimating global sea turtle population sizes, in the first instance census data from index nesting sites are used. Index nesting sites are defined as major nesting areas of known status, or lesser nesting areas for which quantitative data are available. Index nesting sites are generally also ones for which time-series nesting population data are known. According to SEMINOFF (2004a), each index nesting site should be genetically distinct and represent the overall regional subpopulation trends. Moreover the number of individuals at index nesting sites in each region should be proportional to the actual population size in that region. According to the IUCN (2001) *"Subpopulations are defined as geographically or otherwise distinct groups in the*

population between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less)". This definition as used for sea turtle IUCN Red List assessments led to recent discussion, resulting in a reassessment of the Mediterranean *Chelonia mydas*, which was deprived of its unique "subpopulation" status here and delisted from "critically endangered" to "endangered" in the Mediterranean (MROSOVSKY, 2004, 2006). However the "subpopulation" definition used on sea turtles is controversial, as sea turtles show highly migratory behaviour and *C. mydas* males may migrate outside the Mediterranean, whereas females are believed to show higher site fidelity, hence fulfilling the "subpopulation" criteria (NARO-MACIEL and FORMIA, 2006). On the molecular level, different definitions and assessment methods of "genetic exchange" also make a clear subpopulation assessment difficult. As for local populations, assessing the global sea turtle population size based on the number of nesting females at index nesting sites is problematic. First, only females are counted and males are generally excluded from the census. Extrapolations based on an assumed general male/female ratio of 1:1 are critical, considering parameters affecting the pivotal temperature, the constant incubation temperature that produces both sexes in equal measure. This becomes evident taking into account the effects of global warming, shifting this ratio towards females. This was recently described for an important *C. mydas* nesting population at Ascension Island (HAYS et al., 2003). Moreover the female sea turtle nesting population size is subject to natural annual fluctuations, influenced by geographical and climatic factors, namely El Niño and the Southern Oscillation (ENSO). The biological impact of ENSO, associated with cold water and droughts, on the *C. mydas* populations breeding in the Great Barrier Reef, Australia, was described for the first time by LIMPUS and NICHOLLS (1988). In contrast to these Western Pacific populations, Eastern Pacific sea turtle stocks face an opposite effect of ENSO, resulting in increased temperature and heavy rainfall. ENSO events are believed to affect a periodic increase/decrease in sea grass (*Thalassia* spec.) and algae populations, the main food resources for *C. mydas* adults. This may have an impact on the reproductive cycle of this species (MILLER, 1997). Carnivorous species, like *Lepidochelys olivacea* and *Caretta caretta* on the west coast of Baja California, show offshore movement to colder waters, away from the high anomaly areas caused by ENSO. These species may be feeding primarily on pelagic red crabs (*Pleuroncodes planipes*) which prefer the cooler waters. The

correlation between ENSO with its associated climatic events, like hurricanes and monsoons, and changes in sea turtle breeding population sizes, is not yet fully understood and still being investigated. Sea turtle females have remigration nesting intervals and usually lay several clutches in one season. Depending on the species and geographical region these intervals may vary from two up to eight years (CHAN, 2006), and is three years for *C. mydas* and two years for *C. caretta* in the Mediterranean (BRODERICK et al., 2002). Females are also known for skipping breeding seasons, which may depend on the individual's physical condition, size and maturity. Thus the current female nesting population in one region may not represent the total mature female population size (SEMINOFF, 2004a). Clutch sizes and egg numbers within one species and individual are also subject to fluctuations, making conclusions on the female population size, calculated from the number of eggs or nests, difficult. Further problems arise due to the fact that monitoring methods are not yet globally standardized, resulting in data deficiencies and incompatibilities (SWOT, 2007). Many nesting sites may not be discovered due to a lack of monitoring efforts or human resources. This becomes evident in some important sites which were just recently discovered, namely Alata, Turkey, hosting a locally important Mediterranean *C. mydas* population (AYMAK, et al., 2005), Libya, holding large *C. caretta* stocks in the Mediterranean (LAURENT et al., 1999), or Ixtapilla, Michoacán, Mexico, hosting a large *L. olivacea* nesting agglomeration on its Pacific coast (MTSG, 2007). Other sites may not be included in census data for political, logistic, financial or other reasons. Also incomplete or false data, based on wrong counts of female crawls, nesting females, nests or eggs, may result in wrong population size estimation. Inaccurate reading of tags or other marks used for individual identification causes other sources of error. Thus, incorrect monitoring may lead to biased data, highlighting the need for accuracy and awareness when assessing population sizes. It becomes obvious that index nesting site assessment underlies many uncertainties based on lack of information, especially using historic data or small data sets. Using extrapolations of female abundance, which may be highly speculative and lead to bias, causes particular uncertainty. Despite the complexity of problems, the annual female nesting population size estimation at index nesting sites is a standard method for assessing the global status of sea turtles and is applied by sea turtle scientists, research and ecology groups worldwide (SEMINOFF, 2004a, MTSG, 2007, NMFS, 2007).

2.1 Methods

I compared current *Caretta caretta* and *Chelonia mydas* nesting data with light pollution maps, showing the propagation of artificial light in 1996/1997 (CINZANO et al., 2001a). Newer light pollution maps of this type for comparison are not available at present (FALCHI, pers. comm.). First, I evaluated the female nesting population size per season for global index nesting sites. With data providing the number of eggs or nests only, conversions to the number of females per season were made. These conversions are scientifically used for assessing nesting sites (SEMINOFF, 2004a, MTSG, 2007). Data for global index nesting beaches were pooled (Appendix 1). In my thesis the ten largest ("Top ten") nesting areas were subdivided into categories (A-E, Table 2.2) for summarizing sites of similar nesting population size. These categories should be understood as a quantitative tool. This means a nesting site of lower category, with a relatively small number of females/year, is not assessed as less important than a higher category nesting site, as it still may have high regional importance for a genetic stock. A detailed collection of data was made for index nesting sites in the Mediterranean.

Literature research on global nesting sites

For obtaining current nesting site data, literature research was done from December 2004 to November 2007 in the OPAC academic search engine of the University of Bonn, BIOSIS, ISI Web of Science, and Google Scholar. Comprehensive literature research was also conducted in the Marine Turtle Newsletters[5], which have been available since 1976 and updated on a quarterly base, the International Sea Turtle Society Symposium Proceedings[6], which were available from 1988 to 2006, and the Sea Turtle Online Bibliography[7]. The Cturtle mailing list and discussion forum[8] was contacted for information exchange with turtle researchers worldwide. Electronic sources used are the UNEP-WCMC Marine Turtle Interactive Mapping System[9], the State of the World's Sea Turtles (SWOT)[10], and the Global Register of Migratory

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[5] www.seaturtle.org/mtn/

[6] www.seaturtle.org/ists/proceedings

[7] http://accstr.ufl.edu/biblio.html

[8] http://lists.ufl.edu/archives/cturtle.html

[9] http://stort.unep-wcmc.org/imaps/indturtles/viewer.htm

[10] www.SeaTurtleStatus.org

Species (GROMS) database[11]. Reference sources, compiling global nesting data, are SEMINOFF (2004a) for *C. mydas* and SWOT (2007) for *C. caretta*. Sources, focussing on nesting data in the Mediterranean, are KASPAREK (2001) and VENIZELOS et al. (2005) for *C. mydas* and *C. caretta* in the entire Mediterranean, MARGARITOULIS et al. (2003) for *C. caretta* in Greece, and CANBOLAT (2004) for *C. caretta* in Turkey.

2.1.1 Index nesting site compilation

The compilation of index sites in this thesis is based on the number of nesting sea turtle females in one area, defined as nesting population size. Nesting data were compiled for geographical regions that are known for sea turtle nesting activity. Information on species, name of the nesting beach or area assessed, geographical position of nesting beach (latitude/longitude), number of eggs, nests and/or females per nesting season (depending on availability of data), and data source (author) was entered into a database (see Appendix 1). A ranking was made comparing the annual female population size per country. Since this thesis focuses on sea turtle conservation, population size compilation per country, as per administrative unit, was found reasonable. Though compiling current population sizes, population trends were also taken into consideration, depending on the availability of data.

Conversion parameters: clutch size (CS) and nesting frequency (NF)

Since several literature data do not include information on female population size per season, conversions were made for calculating female population size from the number of eggs per nest (clutch size, CS) and number of nests per female per nesting season (nesting frequency, NF) for *C. caretta* and *C. mydas*. This step was crucial to achieve uniformity of data for better comparison.

Table 2.1: Conversion parameters for clutch size (CS) and nesting frequency (NF) Conversions for *C. caretta* were made according to DODD (1988), for *C. mydas* according to MILLER (1997).

Species	mean CS	mean NF
Caretta caretta:	100 eggs/nest	4 nests/female/nesting season[1]
Chelonia mydas:	115 eggs/nest	3 nests/female/nesting season

[1] A nesting "season" is understood as a certain time interval in which the nesting occurs, in the northern hemisphere usually the summer months of a year. For species nesting in the southern hemisphere, nesting may be shifted to the winter months and overlap with the next year.

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[11] www.groms.de/

Examples

1 a) 10 000-20 000 *C. mydas* <u>nests</u> per nesting season estimated for total Indonesia (HALIM et al., 2001).

Minimum population size: 10 000, divided by 3 (nests/female/nesting season) = 3333 females per season.

Maximum population size: 20 000, divided by 3 = 6666 females per season.

1 b) 1500-2300 *C. caretta* <u>nests</u> per season estimated for the Eastern Yucatan Peninsula (TEWG, 2000).

Minimum population size: 1500, divided by 4 (nests/female/nesting season) = 365 females per season.

Maximum population size: 2300, divided by 4 (NF) = 575 females per season.

2) In total, 6 727 400 *C. mydas* <u>eggs</u> were laid between 1984 and 1989 on Tawi-Tawi Turtle Islands, Philippines (TRONO, 1991).

Average population size: 6 727 400 eggs divided by 6 (years) = 1 121 233 eggs/year, divided by 115 (eggs per nest) = 9750 nests/year, divided by 3 (NF) = 3250 females per season.

Except in using the conversions described above, no extrapolations were used here. Based on the calculated nesting data, nesting sites per country were categorized for better comparison of population sizes and mapping. The "Top ten" *C. caretta* and *C. mydas* global nesting sites, as per nesting females/year, were compiled per countries (Figures 2.2, 2.3).

Table 2.2: Global *Caretta caretta* and *Chelonia mydas* nesting site (Top ten) categorization

Category **A**	> 10 000 females/year
Category **B**	5000-10 000 females/year
Category **C**	1000-5000 females/year
Category **D**	500-1000 females/year
Category **E**	< 500 females/year

For setting a focus on European species conservation, *Chelonia mydas* and *Caretta caretta* nesting sites in the Mediterranean were compiled in detail, summarizing all index nesting beaches for which newer data (1990-2004) was available (Figures 2.4, 2.5). To maintain uniformity of the data (provided as nests/year), which allowed good comparability, conversion to "females/year" was not done here.

2.1.2 Data entry in ArcView GIS

Georeferenced nesting data for both species (Appendix 1) were imported into ArcView GIS. If not specified in the literature used, geographical position (latitude/longitude) of the nesting beaches was determined using the open source software Google Earth and entered in ArcView 3.2. All data points entered were also counter-checked visually at high zoom level, comparing location of data points with the "country02"-shapefile. Data points positioned off the coastline were corrected. For global index nesting sites (Chapter 2.2.1), accuracy of data points was highly variable. Some literature data did not specify the exact position of a nesting beach but only general nesting distribution (e.g. Yucatan Peninsula; TEWG, 1998, 2000). Therefore for global nesting sites only descriptive statistics were made for comparison of the estimated annual female population size per country. For sites providing long-term monitoring data, population trends were also taken into consideration (Figs. 2.2, 2.3). Other statistical evaluation was not found to provide reliable and representative outcomes here. In contrast, most Mediterranean sites (Chapter 2.2.2) provided detailed information on the location of the nesting beach (e.g. "Zakynthos, Laganas Bay, Greece") and allowed allocation of coordinates in ArcView with a positional accuracy of 0.001 decimal degrees (± 80 m). These data were used for a SPEARMAN's rank correlation test, comparing light pollution levels in 1996/1997 with contemporaneous nest numbers available from 1990-2004 (see Chapter 3.2.2). Many Mediterranean index nesting sites, while providing time series, are lacking long-term monitoring data (> 10 years). This is regarded as a possible source of error for assessing population trends here. The Mediterranean also shows strong natural annual fluctuations in nesting population sizes (Figures 2.1 a, b).
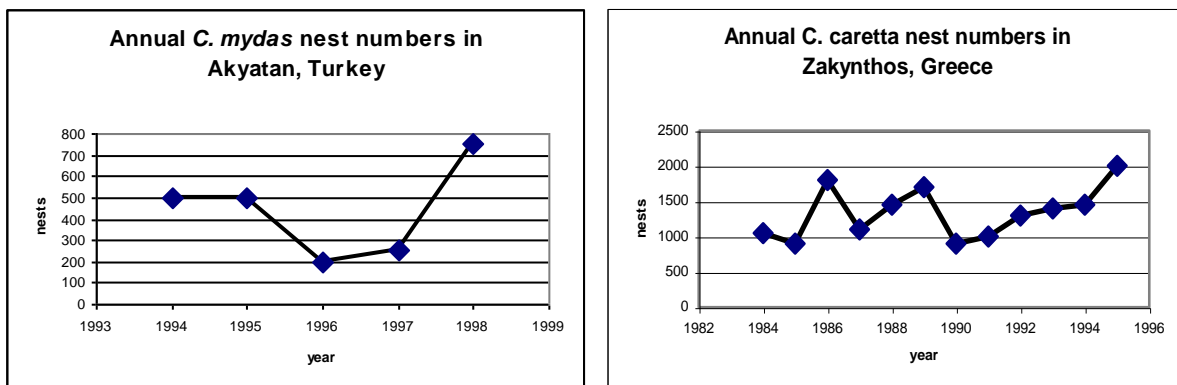
Figures 2.1 a, b: Annual fluctuations in the Mediterranean nesting populations assessed at the primary nesting grounds Akyatan, Turkey (*C. mydas*) and Zakynthos, Greece (*C. caretta*) for which time series data are available. Data are taken from BRODERICK et al. (2002).

2.2 Results

2.2.1 Top ten global nesting sites

In the following, the current Top ten nesting sites are described for the Loggerhead (*Caretta caretta*) (Figure 2.2) and the Green Turtle (*Chelonia mydas*) (Figure 2.3)

Caretta caretta (Linnaeus, 1758)

Highest *C. caretta* nesting occurs in **Oman** with an estimated 30 000 females nesting per year (UNEP-WCMC, 2005, SWOT, 2007) in recent years. Although there is a lack of continuous long-term monitoring data for this region, 30 000 females/year in the late 1970s indicate that this "category A" population is at a stable level.

The **United States of America** hosts the second-largest *C. caretta* nesting aggregation in the world and the largest in the Atlantic (MEYLAN et al., 1995). The Florida Coastline from Melbourne Beach to Wabasso Beach is one of the main nesting areas within the United States, with 36 346 nesting females/year (WEISHAMPEL et al., 2003). For Melbourne Beach there was an increase of 7652 females/year from 1988 to 1992. Other index nesting sites also indicate an increase in population trends on Florida's beaches (TEWG, 2000). Smaller nesting occurs in South Carolina, Georgia and North Carolina (TEWG, 2000, SWOT, 2007). The total annual nesting population size in the south-east USA is estimated to be 20 000 to 30 000 females/year, which is "category A".

Australia holds two *C. caretta* stocks, divided into the Western Australian and the Eastern Australian (Queensland) stock. Traditionally Shark Bay and Dirk Hartog Island host the largest populations of this species in Western Australia, but there is no actual nesting data available for this region (SWOT, 2007). The UNEP-WCMC (2005) gives an estimation of 1000-5000 females nesting on Dirk Hartog Island annually in 1999. Further nesting grounds include Murion Island and Cape Range, Ningaloo, with each area hosting 500-1000 females in 1999 (UNEP-WCMC, 2005). Although there is a lack of long-term census data for any index beach in Western Australia from which population trends can be assessed (LIMPUS, 2002), the Western Australian nesting population is believed to be about an order of magnitude greater than the Eastern Australian population (BALDWIN et al., 2003). Nesting sites in Eastern Australia providing long-term census data are Capricorn and Bunker Islands and Wreck Island, with an estimated past nesting population of 1000 females/year each (BUSTARD in ROSS, 1979). More recent data indicate a decline in the nesting

population for the whole Eastern Australian Region from 3500 to 500 females/year from 1970 to 2003 (LIMPUS and CHATTO, 2004). Current nesting data for Wreck Island, Mon Repos and Heron Island confirm a decrease in the Eastern Australian nesting population (SWOT, 2007). Taking into account the strong decrease in the Eastern Australian stock, the total annual Australian nesting population size is 5000-6000 females/year, with "category B" status.

There is no long-term census data available for **Cape Verde** but newer monitoring data confirm that this *C. caretta* population is the largest in Africa and also of global importance. The total annual Cape Verde nesting population on Boa Vista is indicated to be 3121 females/year (SWOT, 2007), which is "category C".

For **Brazil** there is long-term monitoring data available since 1982, illustrating that the Brazilian nesting population is one of the largest in the world. Main nesting beaches are located in the state of Espirito Santo (BAPTISTOTTE et al., 2003), Praia do Forte, Bahia, (MARCOVALDI and LAURENT, 1996), Maranhao and Ceara (MARQUEZ, 1990). The total nesting population of the states of Rio de Janeiro, Espirito Santo, Bahia and Sergipe is indicated to be 5285 nests (SWOT, 2007). The total current Brazilian nesting population is estimated to be 1000-2000 females/year, with "category C" status.

The Eastern Mediterranean *C. caretta* population is of global importance. Nesting occurs mainly in **Greece**, which holds the largest nesting assemblage in Laganas, Bay, Zakynthos (MARGARITOULIS, 2005, MARGARITOULIS, 2000, MARQUEZ, 1990) and in Kyparissia Bay (MARGARITOULIS and REES, 2001). The second-largest nesting area in the Mediterranean providing time-series for this species is Belek, **Turkey** (CANBOLAT, 2001). Nesting also occurs in Cyprus, Libya, Egypt and Lebanon. The whole nesting population in the Mediterranean was an estimated 2280-2787 females/year in the period 1995-2000 (BRODERICK et al., 2002), which is "category C". Newer estimates are not available at present.

Long-term monitoring data are provided for **South Africa**. The nesting population in Tongaland, KwaZulu-Natal was estimated to be about 500 females/year in the 1970s (HUGHES in ROSS, 1979). Newer monitoring data confirm that this population is stable or increasing. UNEP-WCMC (2005) data provide a population size of 600 to 1000 females in Sodwana nesting annually from 1963-1997. More up-to-date data give an estimation of 238 females nesting on several beaches in KwaZulu-Natal annually (SWOT, 2007). This is "category E" status.

In the Pacific Ocean, **Japan** hosts a larger *C. caretta* nesting agglomeration. Thirty percent of all nesting occurs on the Yakushima islands, on the Maehama and Inakahama beaches, with 100-500 females/year each (UNEP-WCMC, 2005). The total number of nests on these beaches was 1758 in 2005 (SWOT, 2007). Latest monitoring data suggest that this population is stable at about 500 nesting females/year, which is "category D".

The Caribbean coast of **Mexico** hosts an important nesting *C. caretta* population. Census data from the 1970s indicate that there was a population size of about 500 females/year in the state of Quintana Roo (MARQUEZ in ROSS, 1979). There is evidence that the population size in the Eastern Yucatan Peninsula is in a slow decline. 1500-2300 nests/year were counted in the early 1990s (TEWG, 1998), but only about 1000 nests/year in 1998 (TEWG, 2000). Current monitoring data provide a number of 961 nests/year for 2005 (SWOT, 2007). Considering the monitoring data available, it is estimated that the annual population size does not exceed 250 females/year, hence "category E".

Smaller *C. caretta* nesting agglomerations of historic importance are summarized in ROSS (1979). In the 1970s, Santa Marta, Colombia, hosted 400 females/year (KAUFMANN, 1973), Paradise Islands, Mozambique had 300 females/year (HUGHES, 1974). Fort Dauphin, Malagasy Republic, hosted 300 females/year (HUGHES, 1974). All these populations have declined since the 1970s.

Figure 2.2: Current distribution of Top ten *Caretta caretta* nesting sites, as per female nesting population per country. Categories (A-E) indicated according to Table 2.2. Arrows indicate population trends. Preliminary nesting data for Libya (LAURENT et al., 1995, LAURENT et al., 1999) is not included.

***Chelonia mydas* (Linnaeus, 1758)**

According to the IUCN Red List assessment (SEMINOFF, 2004a), the Eastern Pacific Green Turtle (*Chelonia mydas agassizi*) is listed as a subspecies of *C. mydas* and is included in this section.

Australia hosts one of the largest *C. mydas* agglomerations globally. The UNEP-WCMC (2005) gives a rough estimation for Western Australia of 10 000-100 000 females nesting annually on each of Lacepede and Barrow Islands. Moreover, 1000-5000 females/year nest on Murion Island, Cape Range, and the Dampier Archipelago, in each case. There is no detailed data available to confirm these estimations. According to LIMPUS in SEMINOFF (2004a), Heron Island and Raine Island represent the most important nesting areas in the Southern and Northern Great Barrier Reef, Queensland, with 18 000 females nesting on Raine Island in 2001 alone. LIMPUS and LIMPUS (2005) give a higher estimation of 40 000 females nesting on Raine Island annually, but point out that this population may be in a serious decline within the next decades. Considering data provided by SEMINOFF (2004a) and LIMPUS and LIMPUS (2005), and the rough estimations for smaller nesting agglomerations in this region (UNEP-WCMC, 2005), the annual total population size of Queensland is estimated to be about 40 000 females/year. This Eastern population alone qualifies Australia as a "category A" *C. mydas* nesting site.

Tortuguero, **Costa Rica**, is one of the best-known *C. mydas* nesting sites monitored since the 1950s and also one of the largest in the world. Monitoring data indicate that this Caribbean population was increasing steadily from about 8333 nesting females in 1975 to 21 952 – 23 522 females in 2002 (SEMINOFF, 2004a). TROENG and RANKIN (2005) estimate a mean population size of 17 402 - 37 290 females/year from 1999-2003. Due to these positive trends, the annual population size is estimated to be 20 000 to 30 000 females/year, so "category A".

In the Indian Ocean, **Oman** hosts the largest *C. mydas* population, with 10 000-100 000 females per season nesting in Ras Al Hadd (UNEP-WCMC, 2005). Different data indicate that this number should be adjusted downwards to 6000 - 18 000 females/year (SALM, 1991). According to SEMINOFF (2004a), Oman's population size is stable with 6000 females/year nesting between 1979 and 1988. Taking these population trends into consideration, it is estimated that the annual population size is not exceeding 6000 females per season nowadays, which is "category B".

The **Comoros Islands** north of Madagascar hold another important *C. mydas*

nesting population in the Indian Ocean. Monitoring data compiled by SEMINOFF (2004a) indicate that there was a strong nesting population size increase from the early 1970s from 1850 females/year in 1973 to 5000 females/year in 2000. Though there is no newer census data available, it is estimated that this population is still increasing or stable with 5000 females/year, hence "category B or C".

Indonesia holds a large number of *C. mydas*, mainly on the Berau Islands, located east of Borneo. The total number of nests made by this species was assessed at 10 000 - 20 000 per season in all of Indonesia (HALIM, 2001). This accounts for a total nesting population size of 3333 - 6666 females/year in Indonesia. These numbers correspond with average population size data, indicating 1000 - 5000 females/year on each of the two main nesting beaches, Pulau Bilang, Bilangan and Sangalaki, Berau Islands (UNEP-WCMC, 2005). Long-term census monitoring data confirm that there was a strong population size decline on index beaches in the Berau Islands, from 36 000 females/year in the 1940s to 4500 females in 1984 (SEMINOFF, 2004a). Nesting in Indonesia also occurs in Pangumbahan, West Java, and Suka Made, East Java. These sites experienced declines. Due to the overall negative population trends the annual population size in Indonesia is estimated to be < 5000 females/year, which is "category B to C".

On the **Seychelles,** located northeast of Madagascar, the Aldabra Atoll is a major index nesting site for *C. mydas*. It is estimated that 1000 - 5000 females nest annually on its beaches (UNEP-WCMC, 2005). Like the Indonesian stock, this stock also experienced a strong population decline within one century, from 12 000 females/year in 1900 to 4145 females/year in 1996 (SEMINOFF, 2004a). Thus there is evidence that the annual population size is below 5000 females/year, so "category C".

Ascension Island, **St. Helena**, United Kingdom, located in the South Atlantic between South America and Africa, hosts one of the largest agglomerations of nesting *C. mydas* in the Atlantic. Monitoring data indicate that there was an increase in population size from the late 1970s on. A 13 881 nests were estimated in the 1998/1999 season (GODLEY et al., 2001), representing 4627 females in this period. These numbers correspond roughly with the estimation of 3709 females/year in 2001 (SEMINOFF, 2004a). It is estimated that this stock is stable or increasing with 4000-5000 females/year, which is within "category C".

Malaysia, once hosting large *C. mydas* rookeries, has undergone a serious decline

in most of its nesting populations. Nesting in large numbers still occurs on Sabah Turtle Islands, located near the northeast coast of Borneo. The UNEP-WCMC (2005) estimated the total *C. mydas* population of Sabah Turtle Islands to be 1000 - 5000 females in 2001, with index nesting beaches located at Ti Bakkungan Kechil, Ti Selingan and Ti Gulisan. SEMINOFF (2004a) gives a population size estimation of 3251 females/year for Sabah. Historical index nesting sites are Sarawak, located on the north coast of Borneo, with 7549 females/year estimated between 1927 and 1934, and Peninsular Malaysia, with 3096 females nesting here in 1961. Currently both sites do not exceed a number of 1000 females/year (SEMINOFF, 2004a). Due to the overall long-term negative trend for the total Malaysian population, it is presumed that at present < 5000 females/year nest in this area, which is "category C".

Nesting on the **Philippines** mainly occurs on Ti Taganak, located near the Malaysian coast, with 600-1000 females/year, and Ti Baguan, Tawi-Tawi Turtle Islands, with 1000-5000 females/year (UNEP-WCMC, 2005). TRONO (1991) indicates a total nesting population of 1000 females/year on the Tawi-Tawi Turtle Islands. Long-term census data suggest that the *C. mydas* population in the Philippines has undergone a decline from 4886 females/year in 1951 to 3891 females in 1981-1985. Taking the continuous decline in the Philippine stocks into account, it is assumed that the annual population size does not exceed 3000 females/year. This is "category C".

Brazil is believed to have a stable population size with 3000 females nesting annually on Isla Trinidade from 1981 to 2000 (SEMINOFF, 2002). This is "category C".
C. mydas nesting with 1000-2500 females/year also occurs in the Eastern Atlantic Ocean in Guinea-Bissau (CATRY et al., 2002), in the Western Atlantic Ocean in Surinam (LUKE et al., 2004) and the Yucatan Peninsula, Mexico (MARQUEZ, 1990), in the Western Indian Ocean on Tromelin and Europa Islands (LAGARDE et al., 2001, RENE and ROOS, 1996), and in the Eastern Pacific Ocean on the Galapagos Islands, Ecuador (HURTADO, 2001). Nesting aggregations of < 1000 females/year are found in Michoacán and Playa Cuixmala, Jalisco, Mexico (GARCIA et al., 2003). The Michoacán stock of the Eastern Pacific *C. mydas* experienced extreme population declines from 25 000 to 1400 females/year between 1982 and 2001 (CHASSIN-NORIA et al., 2004). Smaller *C. mydas* nesting with < 500 females/year occurs in Hawaii (BALAZS and CHALOUPKA, 2004) and Florida, USA (WEISHAMPEL et al., 2003), Bioko, Guinea (CASTROVIEJO et al., 1994), Aves Islands, Venezuela (LUKE et al., 2004), in

the Mediterranean, mainly in Cyprus and Turkey (KASPAREK et al., 2001, CANBOLAT, 2004), as well as on Wan-An Island, Taiwan (CHEN and CHENG, 1995).
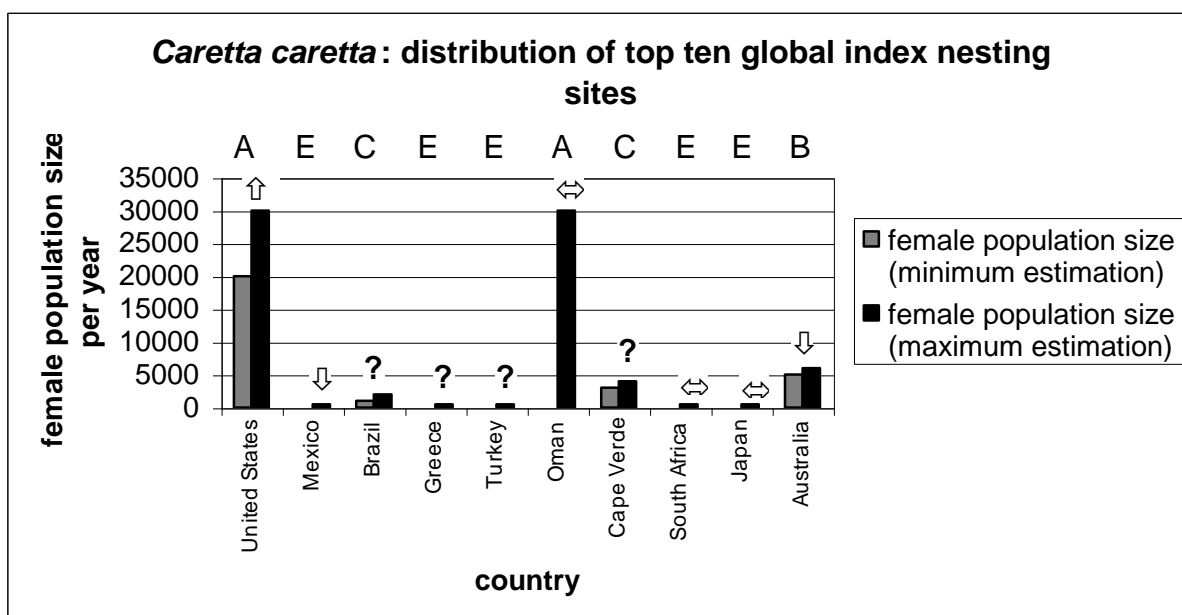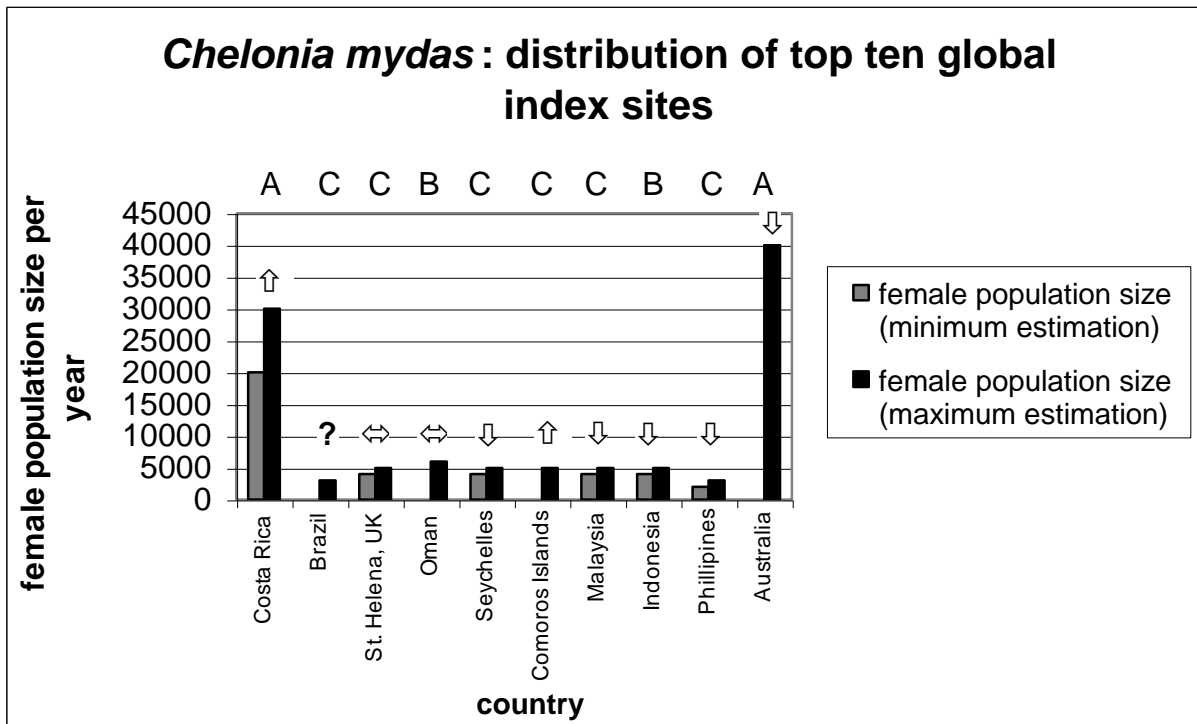

Figure 2.3: Current distribution of Top ten *Chelonia mydas* nesting sites, as per female nesting population per country. Categories (A-E) indicated according to Table 2.2. Arrows indicate population trends.

2.2.2 Mediterranean index nesting sites

In the following, the Mediterranean index nesting sites of *Caretta caretta* and *Chelonia mydas* are described. These are the only species nesting in the Mediterranean regularly.

Loggerhead (*Caretta caretta*) population size in the Mediterranean

According to BRODERICK et al. (2002) there is a nesting population of 2280-2787 *C. caretta* females per year in the entire Mediterranean. Nesting mainly occurs in Greece, Turkey and Cyprus. The largest nesting site in the Mediterranean, providing long-term census data, is Laganas Bay at Zakynthos, Greece, with about 1301 nests/year (MARGARITOULIS et al., 2003). This accounts for about 25% of the total documented *C. caretta* nesting in the Mediterranean (MARGARITOULIS, 2005). Kyparissia Bay, Greece, is considered to be the second-largest nesting area in the Mediterranean (MARGARITOULIS and REES, 2003), with about 620 nests/year. Major nesting in Greece also occurs in Lakonikos Bay, with 192 nests per year. In Crete, index nesting sites are found in the Bay of Chania, with 115 nests/year and Rethymno, with 387 nests/year (MARGARITOULIS et al., 2003). According to CANBOLAT (2001), Belek, Turkey, is the second-largest *C. caretta* nesting site in the entire Mediterranean, with 647 nests per year. Hence this site is equal in female population size with Kyparissia Bay. Further index nesting sites in Turkey are Dalyan, Dalaman, Fethiye, Patara, Kale, Kumluca, Kizilot, Demirtas, Anamur and the Göksu Delta, which hold major *C. caretta* nesting agglomerations (CANBOLAT, 2004). In Cyprus, Chrysochou Bay in the south, and Lara and Toxeftra located on the west coast, are major *C. caretta* nesting sites with 120 and 63 nests counted per year respectively. At Alagadi, Northern Cyprus, 63 *C. caretta* nests are deposited per year. In North Africa, Libya turns out to be a major nesting area for this species, with 9000 nests per season, estimated for the whole country. This extremely high number is based on census data for the eastern part of the country, using extrapolations for the north-western coastline stock, which was unknown at that time (LAURENT et al., 1995, LAURENT et al., 1999). However these data need verification, which will be discussed further. Egypt holds smaller *C. caretta* populations to the east of Port Said, North Sinai. In the Middle East there is evidence of nesting reported for Lebanon. Despite increasing efforts for area-wide monitoring, long-term census data for Libya, Egypt and Lebanon are still lacking (VENIZELOS et al., 2005).
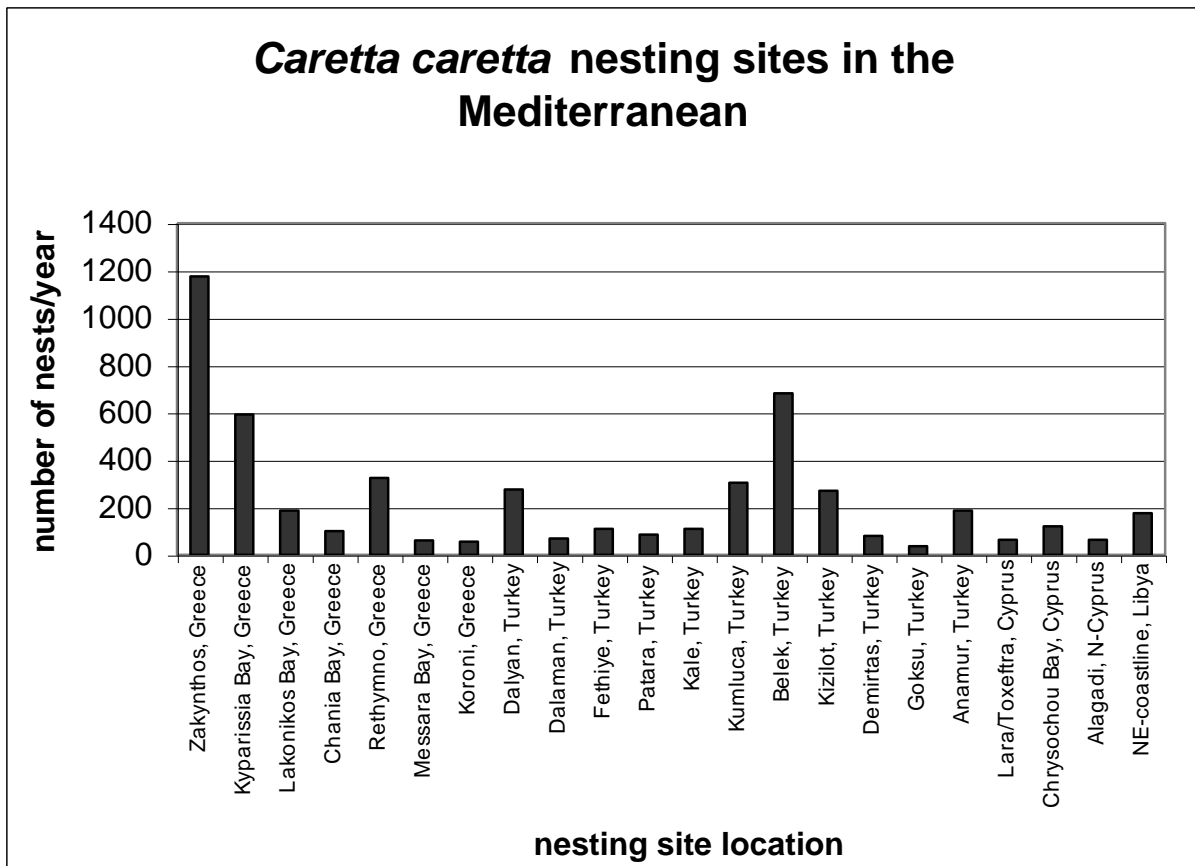
Figure 2.4: *C. caretta* nesting sites in the Mediterranean, providing current nesting data (1995-2002), listed per country. Note the important status of Zakynthos and Kyparissia Bay (Greece) and Belek (Turkey) for this species. Data provided for Libya probably does not reflect the effective population size in the whole country (LAURENT et al., 1999). All data are considered annual average values.

Green Turtle (*Chelonia mydas*) population size in the Mediterranean

The *C. mydas* population in the Mediterranean is considerably smaller than the *C. caretta* population, with about 115 to 580 females depositing 350-1750 nests per year (KASPAREK et al., 2001); 339 to 360 nesting females per year are estimated by BRODERICK et al. (2002). The main nesting areas are Turkey and Cyprus. In Turkey, five regions are major index-nesting sites. Akyatan holds the largest nesting beach, with 353 nests annually (CANBOLAT, 2004). Smaller in size is Sogözü Beach at Yumurtalik, with 213 nests/year (CANBOLAT et al., 2005) and Kazanli, with 156 nests/year (CANBOLAT, 2004). Alata, a recently discovered important nesting ground of this species, holds 128 nests/year (AYMAK et al., 2005). Samandag records 84 nests per year at its main nesting beach Seyhhidir (CANBOLAT, 2004). The Akamas Peninsula in the western part of Cyprus holds 50 nests/year at Lara and Toxeftra (DEMETROPOULOS and HADJICHRISTOPHOROU, 1995). Northern Cyprus hosts important nesting agglomerations at Alagadi, with 68 nests/year (BRODERICK et al., 2002) and at North Karpaz, with 104 nests recorded annually (KASPAREK et al., 2001). *C. mydas* nesting is also confirmed for Syria, with 104 nests/year located at Jablah and Latakia (REES et al. 2005).
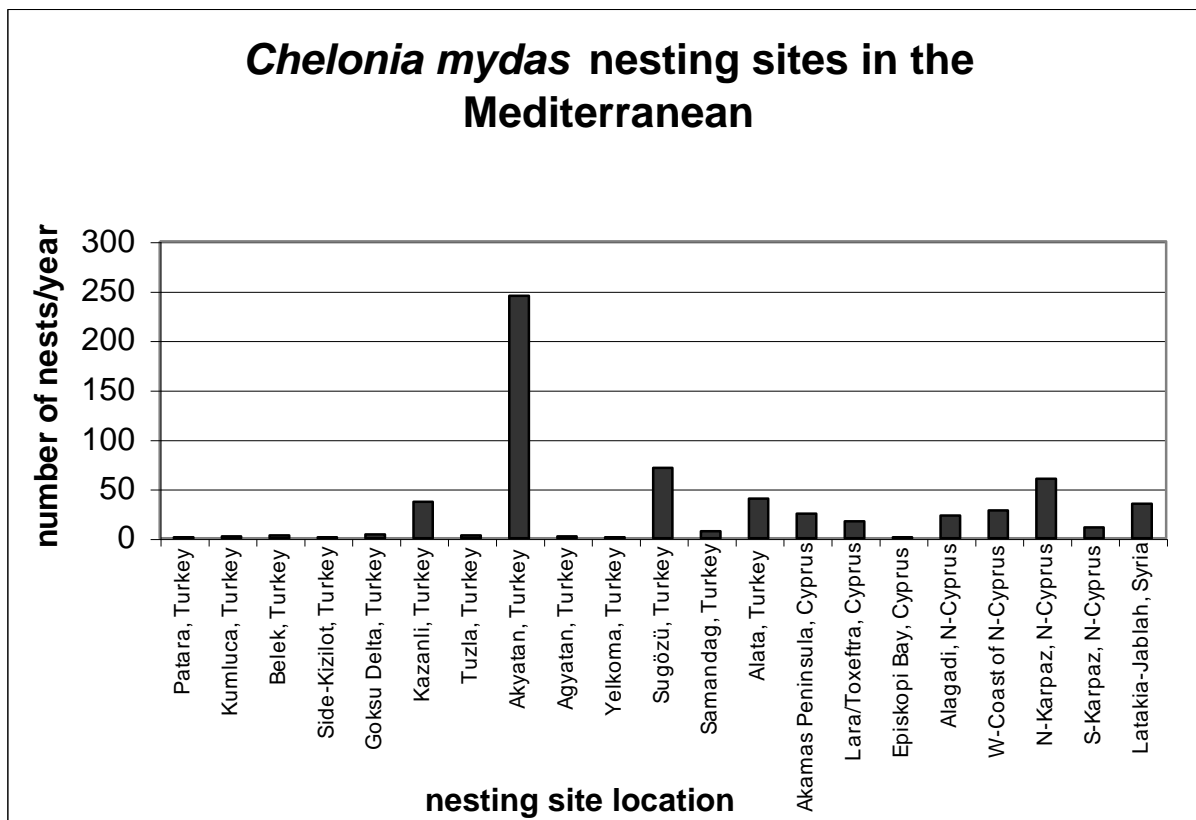
Figure 2.5: *C. mydas* nesting sites in the Mediterranean, providing current nesting data (1990-2004), listed per country. All data are considered annual average values. Note importance of Akyatan (Turkey) for this species.

2.3 Discussion

Estimations of the effective global sea turtle population sizes are complicated. This is because generally adult females are counted on their nesting beaches (SEMINOFF, 2004a, MTSG, 2007, NMFS, 2007), whereas males are excluded from many surveys. Further problems arise because the monitoring methods applied on nesting beaches are not yet standardized globally, making impartial estimates of female population sizes difficult too. Assessing population trends may also be speculative when using incorrect data from the past, or extrapolations from recent data (MROSOVSKY, 2004). Therefore all data given in this chapter should be understood as approximations of effective nesting population sizes. Starting my thesis in 2005, I faced the problem that there were recent data available for *C. mydas* (SEMINOFF, 2004a) but for *C. caretta* data were more incomplete. This lack of knowledge is now partly compensated by the SWOT database, which provides the latest monitoring data also for *C. caretta* index sites (SWOT, 2007). Nevertheless, the outdated IUCN assessment from 1996 (see Table 1.1) and ongoing data deficiency for *C. caretta* highlight the urgent need for more up-to-date and accurate monitoring data for correct categorization of this species.

According to SEMINOFF (2004a), a total of 32 *C. mydas* index nesting sites, representing 26 countries, were used for the IUCN Red List assessment in 2004. Currently, this is an up-to-date sea turtle Red List assessment, rating this species as "Endangered" (IUCN, 2004). This categorization is based on an estimated 48% to 67% reduction of the global annual female population size, from an estimated 173 400 to 90 400 (minimum decrease) or 266 100 to 88 400 individuals (maximum decrease) within the last 3 generations. According to SEMINOFF (2004a), the average generation length for *C. mydas* is 35.5 to 49.5 years, thus 3 generations being equivalent to 100 to 150 years. Based on the literature, I calculated a minimum 94 000 females/year at the Top ten *C. mydas* nesting sites (Figure 2.3; Appendix 1 b). This is above the maximum population size of 76 700 females/year for these ten index sites, estimated by SEMINOFF (2004a). The upward deviation in my compilation is caused by a higher estimation for the Eastern Australian population, taking into consideration the latest data for this area provided by LIMPUS and LIMPUS (2005), who assessed up to 40 000 females per year nesting on Raine Island alone. Australia and Costa Rica currently represent the largest nesting areas by far, probably holding up to 50% of the nesting populations at the major global index sites. Due to the global

importance of these sites, special conservation attention is needed here. The Eastern Mediterranean, which holds nesting populations in Turkey, Cyprus and N-Cyprus (KASPAREK et al., 2001, CANBOLAT, 2004), was not assessed a Top ten nesting area in my rating, as it has a total female population size of only 115-580 (KASPAREK et al., 2001) or 339 to 360 females/year (BRODERICK et al., 2002). On a global level this is very low, compared with at least 2000-3000 nesting *C. mydas* females/year at a Top ten nesting site calculated here (see Figure 2.3; Appendix 1 b). However, *C. mydas* had an important status as a subpopulation in the Mediterranean and was regionally listed as "critically endangered" (IUCN, 2004). This categorization was challenged recently, leading to a status reassessment. This decision is still under discussion (Mrosovsky, 2004; Seminoff, 2004b; Mrosovsky, 2006; NARO-MACIEL and FORMIA, 2006). However, even if not fulfilling the status of a "subpopulation" as per IUCN (2001) definition, without doubt *C. mydas* has crucial importance as a regionally threatened species in the Mediterranean. This species' strictly protected status is also included in the Berne Convention, where it is listed under Appendix II. Thus its nesting habitats should be treated as having highest conservation priority.

C. caretta was rated "endangered" in the 1996 IUCN Red List assessment and there is no newer assessment for categorization available to date. This lack of recent data makes comparison of the data compiled in my thesis from reference sources difficult. Based on the literature data available, I calculated a total 69 500 females/year at the Top ten nesting sites (Figure 2.2, Appendix 1 a). Of these, Oman and the USA are top category index nesting sites (TEWG, 2000, SWOT, 2007), with a total of 55 000 females/year. This accounts for 79% of the total population at the Top ten nesting sites compiled in my thesis. Western Australia also holds large nesting agglomerations, but high population declines were recorded for Eastern Australia from the 1970s onwards, which were probably caused by international fisheries offshore (CHALOUPKA and LIMPUS, 2001, LIMPUS and CHATTO, 2004). This highlights the complexity of problems when considering conservation measures for this highly migratory species. The survival of a local population is not dependent on national conservation efforts only, but also on measures that are taken on an international scale. As shown above, *C. caretta* nesting sites in the Mediterranean are of global importance, probably holding up to 4% of the population at the Top ten nesting sites, as calculated here. Therefore, Turkey, Greece and Cyprus, holding the majority of nesting beaches in the Mediterranean (Figure 2.4) and also providing long-term

census data, should have particular conservation priority. Nevertheless, human impacts are affecting these index nesting sites considerably, which will be discussed using the example of Belek (Chapter 4). The main hope for the survival of this species are areas which are not yet affected by coastal development, specifically in Libya. Though Libya does not provide confirmed long-term census data and the preliminary estimation of 9000 nests/year (LAURENT et al., 1995) may be too high (VENIZELOS et al., 2005, CANBOLAT, 2001), this area could be of future importance given the increasing coastal development in Greece and Turkey which is destroying sea turtle nesting space. How sea turtle females, which are known for nest site fidelity (MROSOVSKY, 1983, BOWEN et al., 1992, MILLER, 1997, BRODERICK et al., 2007), may switch to more undeveloped nesting sites in the Mediterranean should be investigated over time.

Despite overall negative trends for sea turtle populations globally there are some populations that are increasing, namely the *C. caretta* stocks in the Southeast USA (TEWG, 2000) and the *C. mydas* stocks in the Central Pacific, Central Atlantic, and Western Atlantic (SEMINOFF, 2004a). This is due to intensive conservation efforts worldwide. The situation for the Mediterranean is uncertain. Population trends were not assessed here, due to a lack of long-term monitoring data for a number of nesting sites and other statistical uncertainties, based on inconsistency of data (change of assessors, monitoring methods, beach lengths evaluated). The naturally high fluctuation of annual nest numbers in this region must also be taken into consideration (Figures 2.1 a, b). Nevertheless, future conservation efforts will also decide the survival of *C. caretta* and *C. mydas* in the Mediterranean. As a first priority this includes the reduction of bycatch, a problem which was well studied in the Mediterranean in previous years (AGUILAR, 1995, LAURENT et al., 2001). The conservation of coastal habitats is also of top priority, but has not been realized adequately in this nesting region (DEMETROPOULOS, 2001). As shown by the negative example of Zakynthos, Greece, coastal development is even evident in protected areas. Overall light pollution is one of the key problems, as it is known to have a negative impact on nesting sea turtles and their hatchlings (WITHERINGTON and MARTIN, 1996). But quantitative data on light pollution at Mediterranean nesting sites are sparse, and comparative data in this region are lacking completely. To what extent sea turtle index nesting sites are exposed to nocturnal light pollution on a global scale, and in the Mediterranean, will be discussed in the next chapter.

3 Sea turtle index nesting sites and light pollution (1996/97)

3.0 Introduction

The negative effects of artificial lights on life systems and sea turtles in particular are well known by now (VERHEIJEN, 1985, MCFARLANE, 1963; PHILIBOSIAN, 1976; WITHERINGTON, 1992a; WITHERINGTON and MARTIN, 1996). The problem of hatchling disorientation, and disruption of female nesting behaviour, was detailed in the general introduction to this thesis (see Chapter 1). Quantitative analysis of light pollution at sea turtle nesting beaches outside the United States is sparse. Therefore, based on the nesting data compiled in Chapter 2, the next step in this thesis was to create special maps, identifying light pollution "hot spots" at global *Caretta caretta* and *Chelonia mydas* nesting sites. Detailed maps were created for the Mediterranean.

Light pollution measuring techniques

Light pollution, also known as photo pollution or luminous pollution, is generally understood as the excess of anthropogenic (artificial) light at night. The terms "light pollution" and "artificial light" will be used in my thesis constantly.
For defining light pollution in this chapter, the scientific definition according to SMITH (1979) was used:

Artificial sky brightness, which is > 10 percent of the natural night sky brightness above 45 degrees of elevation.

This definition was chosen to evaluate the light pollution maps, which are used in my thesis (CINZANO et al. 2001a, with permission). When assessing light pollution levels in the field, problems arise due to the different sensitivity to brightness of human observers' eyes. This phenomenon can be generalized to animal life systems and is based on WEBER-FECHNER's and STEVEN's physical laws.

Equation 3.1: WEBER-FECHNER Law:

$$E = c \cdot \ln \frac{R}{R_0}$$

E: subjective size of the stimulus
R: magnitude of physical stimulus

Equation 3.2: Steven`s Power Law:

$$E = k \cdot R^n$$

E: subjective size of the stimulus

R: magnitude of physical stimulus

n: exponent depending on the type of stimulation (for brightness perception = 0.33)

According to the Weber-Fechner Law and Steven`s Power Law, there is a logarithmic relationship between the physical magnitudes of stimuli and the perceived intensity of these stimuli. This means that to elicit the same change in perception, defined as the just noticeable difference (JND), high intensity stimuli require a greater increase than low intensity stimuli. Thus assessments of light pollution based on visual perception alone are subject to uncertainties and require many observers and repeated measurements for objectivity. For measuring the visibility of celestial bodies within night sky brightness in a given area, the Bortle Scale was introduced by Bortle (2001). This scale allows a quantification and comparison of light pollution levels in different areas. In a nine-level numeric scale, class 1 (colour on scale: black) defines the darkest sky, whereas class 9 (white) is defined the highest level of light pollution. The Bortle Scale is an appropriate tool for astronomers for identifying and comparing the darkness of observing sites, but difficult to handle for amateurs. The colour coding of the Bortle Scale is also used in the World Atlas of Artificial Night Sky Brightness (see Table 3.1). Measuring the Surface Brightness (SB) in magnitudes/arcsec^2 is a technical method used by astronomers for comparing the brightness of different celestial bodies. The apparent magnitude (m) of a celestial body is a measure of its brightness as seen by an observer on Earth. This method is also applied for measuring light pollution: 22.5 magn/arcsec^2 are defined as the darkest measurable value on earth, whereas 170 magn/arcsec^2 is the highest brightness level, which is found in big cities. The SB is dependent on other light sources at the same site, e.g. the moon. Under full moon conditions there is a lower SB for an observed body at the same site than under half moon or new moon conditions. In addition, under constant lighting conditions the SB stays constant and does not decrease with greater distance. Measurements of the SB are used for applications in satellite technology, which will be described in detail here as it is the basic principle of the light pollution maps used in this chapter.

World Atlas of Artificial Night Sky Brightness

The World Atlas of Artificial Night Sky Brightness provides maps that show light pollution on a global scale for the first time (CINZANO et al., 2001a). These maps are based on nocturnal photography of the earth's surface in the years 1996 and 1997, taken by the Operational Linescan System (OLS) of the Defense Meteorological Satellite Program (DMSP) satellite. The OLS sensor is an oscillating scan radiometer with low light visible and Thermal Infrared (TIR) imaging capabilities, and measures the Surface Brightness, referencing light pollution levels to an average natural sky brightness below the atmosphere, corresponding to 21.6 magn/arsec^2. The maps provided are cloud-free composites, calibrated to the photometric V- band (550 nm), at the zenith, in clean atmosphere, with an aerosol coefficient of K=1. The effective ground sample distance (GSD) of the satellite images is 2.8 km. In Geographic Information Systems (GIS) the GSD defines the spatial resolution of the satellite image.
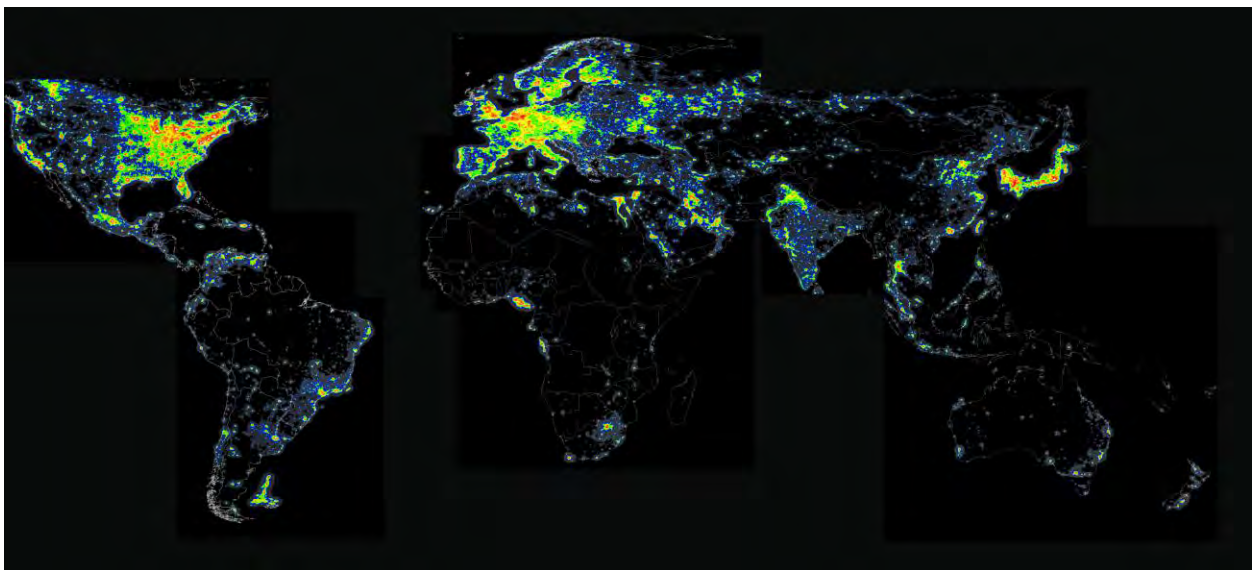
Figure 3.1: World Atlas of Artificial Night Sky Brightness. Maps based on satellite data from 1996/1997 (from CINZANO et al., 2001a, with permission)

Maps from the World Atlas of Artificial Night Sky Brightness were used in this chapter for a number of reasons. First, as the OLS sensor measures light pollution at sea level, these data can be optimally applied for assessing brightness ratios at sea turtle nesting sites. Second, the maps allow a quantification of light pollution, as defined in Table 3.1. Third, the OLS measures the propagation of light in the atmosphere, also taking into account Reighlay scattering by molecules, Mie scattering by aerosols, and atmospheric extinction along a light path and earth curvature (CINZANO et al., 2001a). Therefore it is adequate for covering light pollution area-wide. This is crucial, considering that light pollution at a sea turtle site may not only be caused by irradiation from a close light source, but also by artificial lights or sky glow originating from distant areas. This problem was evident for the nesting beach in Belek, Turkey, what is discussed in Chapter 4 (see Figure 4.11, Appendix 3). Fourth, the OLS has a broad spectral response from 440 to 940 nm, with highest sensitivity between 500 to 650 nm. This range covers Metal Halide lights, with peaks between 550 to 625 nm (Figure 3.2 b), Mercury Vapour lights (554 to 575 nm), High Pressure Sodium Vapour lights (570 to 630 nm) and Low Pressure Sodium Vapour lights (peak at 589 nm). These are artificial light sources also used for outdoor lighting at nesting beaches (see Chapter 1, Table 1.3) and include wavelengths that are disruptive for sea turtles and their hatchlings (WITHERINGTON and MARTIN, 1996, SALMON, 2003).
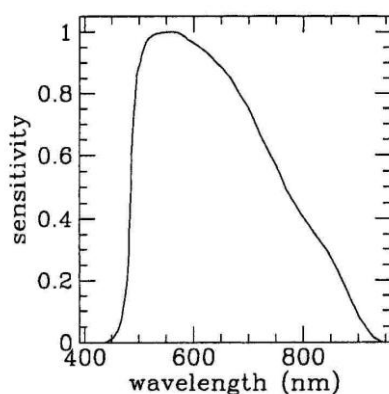
Figure 3.2 a: Spectral sensitivity of the DMSP-OLS (from CINZANO et al., 2000)
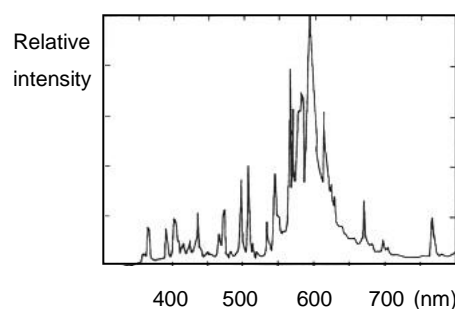
Figure 3.2 b: Spectral emissions of a Metal Halide light source with peaks at 600 nm (yellow).

CINZANO et al. (2001a) used false colour projection in their maps for illustrating different light pollution levels. The false colours correspond to ratios between the artificial sky brightness and the natural sky brightness. In this thesis, these false colours were divided into categories (1-8) for descriptive purposes (Table 3.1).

Table 3.1: Quantification of light pollution in the World Atlas of Artificial Night Sky Brightness. Data according to Cinzano et al. (2001a)

False colour projection and categorization in this thesis	Light pollution ratios
Black "Category 1"	Map area is **< 0.01 times brighter** than natural sky brightness[1]
Grey "Category 2"	Map area is **0.01-0.11 times brighter** than natural sky brightness[2]
Blue: "Category 3"	Map area is **0.11-0.33 times brighter** than natural sky brightness
Green "Category 4"	Map area is **0.33-1 times brighter** than natural sky brightness
Yellow: "Category 5"	Map area is **1-3 times brighter** than natural sky brightness
Orange "Category 6"	Map area is **3-9 times brighter** than natural sky brightness
Red "Category 7"	Map area is **9-27 times brighter** than natural sky brightness
White "Category 8"	Map area is **>27 times brighter** than natural sky brightness[3]

[1] This is the lowest ratio, which corresponds to natural light levels in undeveloped areas.
[2] This category corresponds to the threshold level for light pollution (> *10 percent increase of the natural night sky brightness*), according to SMITH (1979).
[3] This is the highest ratio, which is found in urban core areas only.

3.1 Methods

Based on the sea turtle nesting data from Chapter 2 (see also Appendix 1), nesting distribution maps for *C. caretta* and *C. mydas* were created and superimposed with light pollution maps in ArcView GIS. These high-resolution (2.8 km) maps were kindly approved for usage in my thesis by Dr. Pierantonio CINZANO[12]. For references see CINZANO et al. (2001a), CINZANO et al. (2001b) and CINZANO et al. (2000). To cover global *Caretta caretta* and *Chelonia mydas* index nesting sites, single light pollution maps for Europe, Asia, Africa, Australia and the Americas were downloaded from the First World Atlas of Artificial Night Sky Brightness[13]. For the Mediterranean, light pollution maps for Europe, the Middle East and North Africa were combined to obtain total coverage of its eastern area, which holds the *C. caretta* and *C. mydas* index nesting sites in this region (see Chapter 2).

3.1.1 Superimposition of nesting data with light pollution maps

In a first step, sea turtle nesting site spatial data (latitude/longitude, given in Appendix 1) were entered in ArcView 3.2 using the country02-shapefile for highlighting the coastline. In a second step, the light pollution maps provided in .jpg-format were georeferenced in ArcView 3.2, using the above shapefile. In the georeferencing process the accuracy was defined by comparing the root mean square (rms) of the control points. As lower rms indicate a better coefficient for the spatial match, all rms values above 0.05 were deactivated in the georeferencing process. This resulted in an exact match of the country02-shapefile and the light pollution maps at high zoom levels (see Figures 3.9 - 3.13, 3.15, 3.16). The degree of light pollution at a particular nesting site was assessed by visual inspection of the georeferenced light pollution maps. For global nesting sites (Figures 3.3 – 3.7) an approximation was made to define the light pollution category in a given area. For the Mediterranean, 25 *Caretta caretta* and 24 *Chelonia mydas* nesting sites with accurate spatial data were assigned to one of the eight light pollution categories (Tables 3.3, 3.4). Of these sites, 21 *C. caretta* and 21 *C. mydas* sites provide newer nesting data from 1990 to 2004 (see Chapter 2) and also represent the current index sites in this region. These sites were used for statistical evaluation.

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[12] Dipartimento di Astronomia, V. lo dell'Osservatorio 5, I-35100 Padova, Italy

[13] www.lightpollution.it/worldatlas

3.1.2 Statistical evaluation for the Mediterranean

The proportion of sea turtle nesting sites in the Mediterranean that were exposed to different categories of artificial night sky brightness under standard clear night conditions (see Table 3.1) was calculated. As the maps used in this chapter represent light pollution levels from 1996/1997 (CINZANO et al., 2001a), they allowed a good approximation of the effective nocturnal brightness at sea turtle nesting sites in the period 1990-2004. For nesting sites providing defined spatial information (e.g. Belek, Turkey, see Figure 3.9), the light pollution category at this point was assessed by visual inspection of the georeferenced maps. For nesting sites providing more inaccurate spatial data (e.g. "west coast of Northern Cyprus", Fig. 3.15) or located within different categories of light pollution (e.g. Alagadi, N- Cyprus, see Figure 3.15), the light pollution category was chosen which represented the highest coverage in the given area, which was assessed visually. Consequently, the light pollution category given should be understood as an approximation of existing conditions. The results are given in Chapter 3.2 (Figures 3.8 – 3.16, Tables 3.3, 3.4).

SPEARMAN's rank correlation test

The SPEARMAN's rank correlation test was used to test if high light pollution levels and sea turtle female nesting in the Mediterranean were negatively correlated. It was investigated whether low nesting activity in an area could be used as an indicator for the negative effect of light pollution on the site-choice of females (WITHERINGTON, 1992a, SALMON et al., 1995a).

Both species nesting in the Mediterranean were tested for H_0 or H_1, defined as:

- H_0: There is no correlation between high light pollution levels and low nest numbers.
- H_1: There is a correlation between high light pollution levels and low nest numbers.

Equation 3.3: SPEARMAN's rank correlation

$$\rho = 1 - \frac{6 \sum d_i^2}{n(n^2 - 1)}$$

ρ: SPEARMAN's rank correlation coefficient

d_i: the difference between each rank of corresponding values of x and y

n: the number of pairs of values

The S<small>PEARMAN</small>'s rank correlation test was conducted using the VassarStats statistical computation program[14]. Significance level for H_1: p (two-tailed) < 0.05.

Example

Zakynthos, Greece, had 1301 *C. caretta* nests in 2002 (M<small>ARGARITOULIS</small> and R<small>EES</small>, 2003) and was located within an artificial sky brightness zone of category 5 (yellow), which is up to 3 times brighter than natural sky brightness (C<small>INZANO</small> et al., 2001a).
Resulting observation pairs (x, y): 1301 (nests/year) and 3 (artificial to natural brightness ratio)

The observation pairs (x and y) were defined as:
x: artificial to natural sky brightness ratio in a given area (data taken from Table 3.1)
y: total number of nests at a site for a given year or per season (data taken from Tables 3.3 and 3.4).

Comments: Only sites for which newer nesting data (1990 - 2004) was available were included in these statistics. Dalyan, Alanya and Ayia Napa, known for sporadic *C. mydas* nesting, were not included in statistics, as there was no actual nesting data available for these sites. Oum el Frais, Ras el Aweija, East Sirte and North Benghazi, providing preliminary *C. caretta* nesting data, were excluded from the evaluation as only summarized nesting data for the entire Libyan coastline but no nest numbers per nesting site were available (L<small>AURENT</small> et al., 1997). In total, 21 *Caretta caretta* and 21 *Chelonia mydas* nesting sites in the Mediterranean were evaluated.

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[14] http://faculty.vassar.edu/lowry/corr_rank.html

3.2 Results

3.2.1 Global nesting sites and light pollution

Light pollution maps from 1996/1997 (CINZANO et al., 2001a) reveal that on a global scale sea turtle index nesting sites face highly variable artificial brightness at night (Figs. 3.3 - 3.7). Japan generally shows high light pollution (category 5 or higher) near urban areas in coastal zones (Fig. 3.6), whereas light pollution is considerably lower (category 1 or 2) at the *C. caretta* index nesting site on the Yakushima Islands in the south of Japan. Light pollution is also very high (category 5 or higher) in the Southeast of the United States, in particular on the east coast of Florida, which hosts one of the largest *C. caretta* agglomerations in the world (Figs. 3.4, 3.5). Greece and Turkey, also contributing to important global *C. caretta* sites, both show comparatively high light pollution (up to category 5) in coastal areas (Table 3.3, Figs. 3.9 - 3.12). Moderate-level light pollution ($\pm$ category 3) is evident at nesting sites in Oman and South Africa (Fig. 3.3), Mexico, Costa Rica, Brazil (Fig. 3.4), and sites in Indonesia, Malaysia, the Philippines, Western Australia and South Queensland (Fig. 3.6). No light pollution (category 1) is found in Northern Queensland, namely the Great Barrier Reef, which holds one of the largest *C. mydas* stocks in the world at Raine Island (Fig. 3.7). Light pollution is generally higher in countries which are highly developed and also show high human population density (Tables 3.2 a, b).

Table 3.2 a: Top ten global *C. caretta* nesting sites per country, light pollution (1996/1997) and human population density (1995-2005)

Caretta caretta global Top ten nesting sites indicating nesting population size (category A-E, see Chapter 2)	Percentage of the surface area affected by light pollution in 1996/1997 (CINZANO et al., 2001a)	Increase in human population density (PD) from 1995 to 2005 (United Nations World Population Prospects[15])
Japan (E)	98.5	332 – 338
United States (A)	61.8	28 – 31
Greece (C)	57.7	81 – 84
Turkey (C)	31.2	80 – 93
Mexico (E)	30.5	47 – 53
Oman (A)	27.8	7 – 8
South Africa (E)	13.7	34 – 39
Brazil (C)	7.9	19 – 22
Australia (B)	2.3	2- 3
Cape Verde (C)	No data	99 – 126

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[15] http://esa.un.org/unpp/

Table 3.2 b: Top ten global *C. mydas* nesting sites per country, light pollution (1996/1997) and human population density (1995-2005)

Chelonia mydas global Top ten nesting sites and nesting population size (category A-E, see Chapter 2)	Percentage of the surface area affected by light pollution in 1996/1997 (CINZANO et al., 2001a)	Increase in human population density (PD) from 1995 to 2005 (United Nations World Population Prospects)
Costa Rica (A)	34.1	68 – 85
Oman (B)	27.8	7 – 8
Malaysia (C)	22.2	62 – 78
Philippines (C)	12.6	229 – 282
Brazil (C)	7.9	19 – 22
Indonesia (C)	6.8	104 – 119
Australia (A)	2.3	2 – 3
St. Helena, UK (C)	No data	43 – 52
Comoros Islands (C)	No data	272 – 357
Seychelles (C)	No data	166 – 188
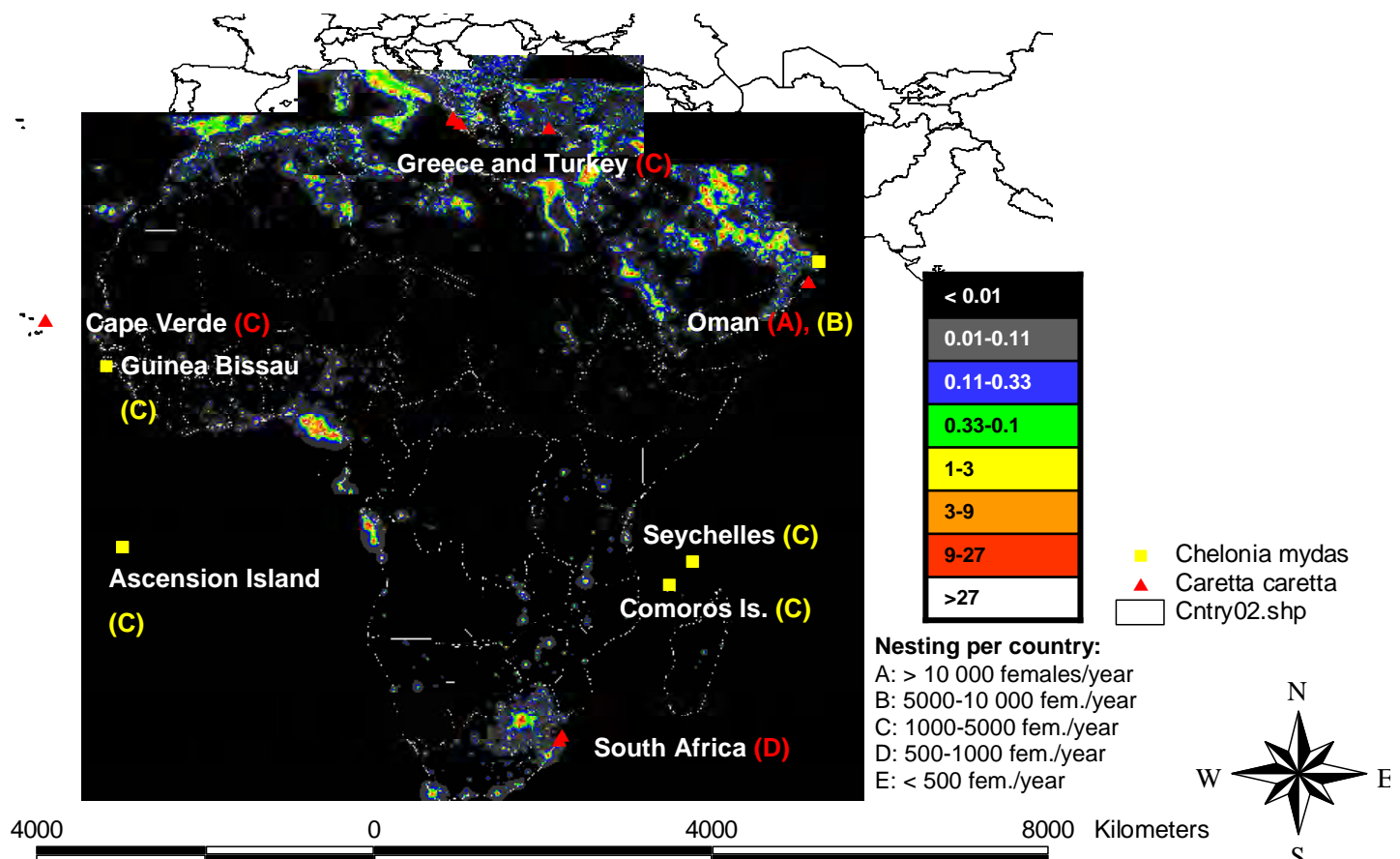
Figure 3.3: Light pollution at Top ten nesting sites in the Middle East, Europe and Africa (1996/1997). Light pollution categories according to Table 3.1. The map highlights light pollution hot spots at *C. caretta* index nesting sites in Greece, Turkey and South Africa, whereas the Masirah Islands, Oman, are affected less. Light pollution is also present in Ras Al Hadd, Oman, which is a *C. mydas* index site. Ascension Island, the Comoros Islands and the Seychelles show no light pollution. No light pollution data are available for the *C. caretta* index nesting sites on the Cape Verde Islands.

Nesting per country:
A: > 10 000 females/year
B: 5000-10 000 fem./year
C: 1000-5000 fem./year
D: 500-1000 fem./year
E: < 500 fem./year

■ Chelonia mydas
▲ Caretta caretta
☐ Cntry02.shp
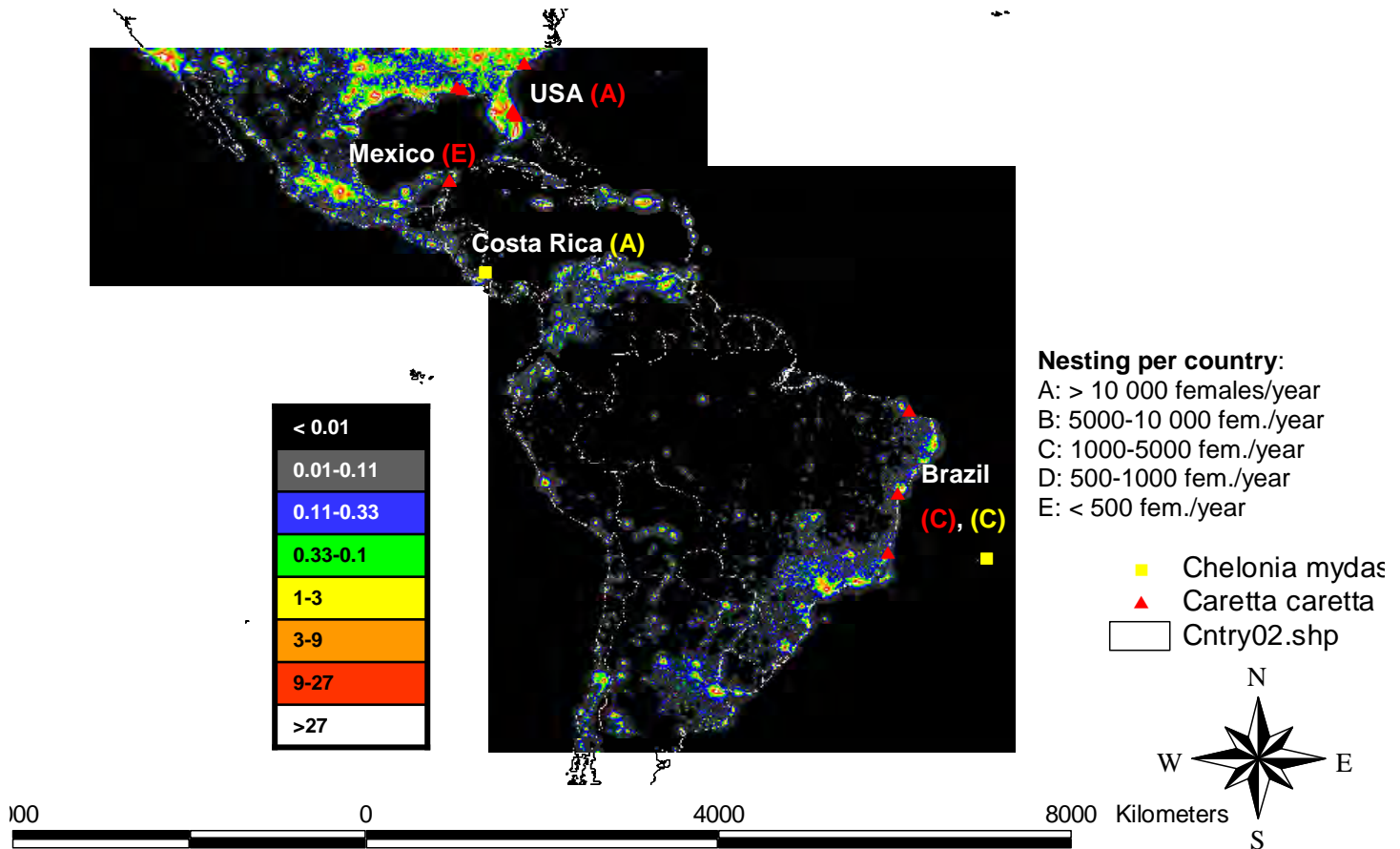
Figure 3.4: Light pollution at Top ten nesting sites in the Americas (1996/1997). Light pollution categories according to Table 3.1. The map highlights light pollution hot spots at *C. caretta* index nesting sites in Florida and South Carolina, United States. Moderate light pollution is evident at *C. caretta* index nesting sites in Quintana Roo, Mexico, and the mainland of Brazil. *C. mydas* nesting sites in Costa Rica and Isla Trinidade, Brazil, face comparably low level light pollution.

Florida, South Carolina
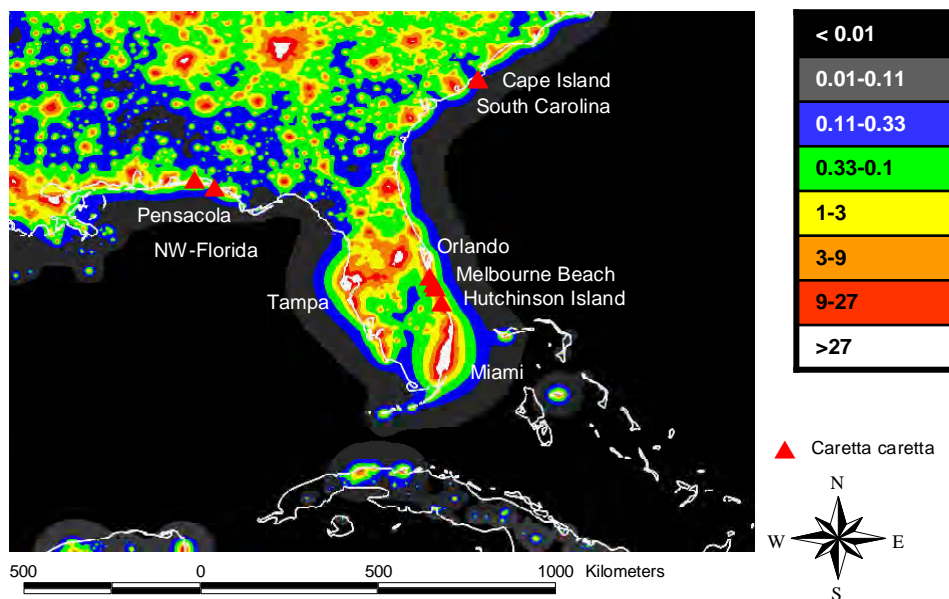
Figure 3.5: Light pollution at *C. caretta* index nesting sites in Florida and South Carolina, southeast USA (1996/1997). Light pollution categories according to Table 3.1. Note high light pollution levels of category 5 (yellow) and higher at *C. caretta* nesting sites in Cape Island, Melbourne Beach, Hutchinson Island and Pensacola as a result of high degree of urbanization. Urban core areas, such as Tampa, Orlando and Miami (Florida) show highest light pollution category 8 (white), with artificial light propagating to adjacent areas.
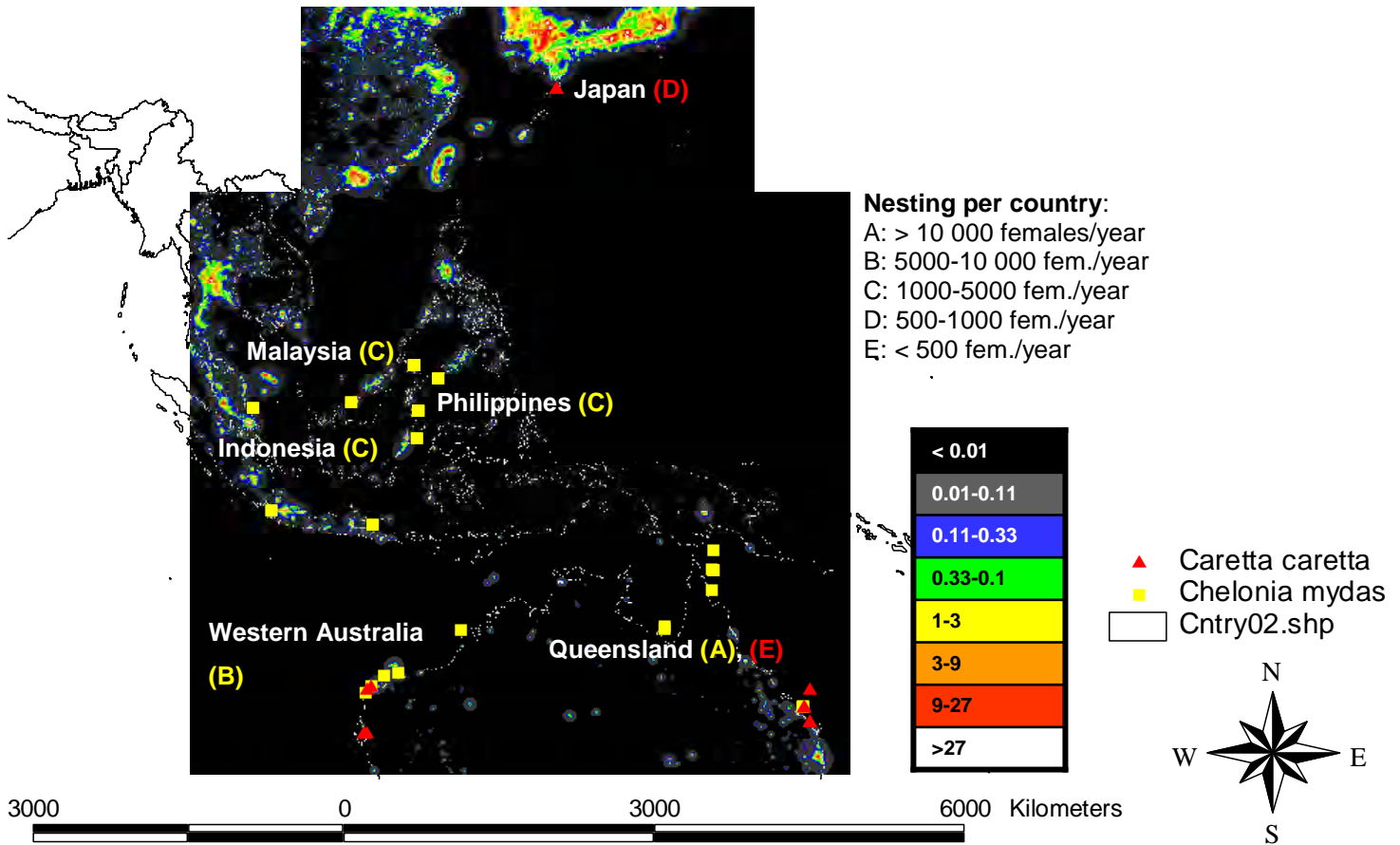
Figure 3.6: Light pollution at Top ten nesting sites in Southeast Asia and Australia (1996/1997). Light pollution categories according to Table 3.1. The map highlights light pollution hot spots near *C. caretta* index nesting sites in the south of Japan. Moderate light pollution is evident at *C. caretta* index nesting sites in Western Australia and the southeast coast of Queensland. *C. mydas* nesting sites in Malaysia, Indonesia and the Philippines also face moderate light pollution near urban areas. Lowest light pollution levels are found at *C. mydas* nesting sites in the north of Australia.

Northern Great Barrier Reef (nGBR)
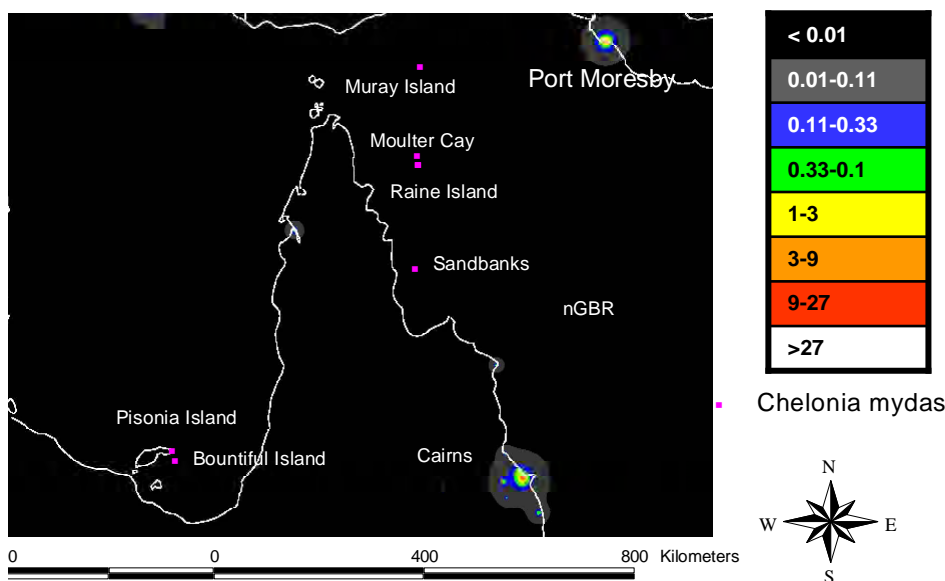
Figure 3.7: Light pollution at *C. mydas* index nesting sites in the Northern Great Barrier Reef, Queensland, Australia (1996/1997). Light pollution categories according to Table 3.1. Note naturally dark nocturnal light conditions of category 1 (black) due to low degree of urbanization in this area.

3.2.2 Mediterranean nesting sites and light pollution

Loggerhead Turtle (*Caretta caretta*)

Superimposition of *C. caretta* index nesting sites with light pollution maps revealed that 76% of 25 nesting sites compiled are located in areas which are above the threshold level of light pollution (> category 2, see Figure 3.8). This is artificial sky brightness which is > 10% of the natural night sky brightness above 45 degrees of elevation (SMITH, 1979).

SPEARMAN's rank correlation test: The outcomes reveal that there is no significant correlation between high light pollution levels and low nest numbers for 21 *C. caretta* index sites assessed [H_1 with p (two-tailed) = 0.08376].

Table 3.3: Mediterranean *C. caretta* nesting sites and exposure to light pollution (1996/1997)

Light pollution category	Nesting site	Latitude/longitude (decimal degrees)	Nest number (year)
1 (lowest level)	Oum el Frais (Libya)	32.194/ 23.265	no data
2	Kyparissia (Greece) Koroni (Greece) Lakonikos(Greece) Patara (Turkey) East Sirte (Libya)	37.277/ 21.683 36.794/ 21.967 36.610/ 22.495 36.317/ 29.245 31.191/ 16.838	593 (2002) 55 (2002) 187 (2002) 85 (2000) no data
3	Dalyan (Turkey) Demirtas (Turkey) Göksu Delta (Turkey) Lara/Toxeftra (Cyprus) Chrysochou Bay (Cyprus) Alagadi (Northern Cyprus) Ras el Aweija (Libya) North Benghazi (Libya)	36.834/ 28.645 36.406/ 32.171 36.297/ 34.032 34.916/ 32.323 35.041/ 32.412 35.334/ 33.490 30.855/ 17.907 32.295/ 20.235	276 (1999) 80 (1996) 36 (1996) 64 (1995) 120 (2002) 63 (2002) no data no data
4	Dalaman (Turkey) Kale (Turkey) Rethymno (Greece) Messara (Greece)	36.690/ 28.765 36.227/ 30.001 35.363/ 24.460 35.011/ 24.763	69 (1998) 109 (1998) 325 (2002) 61 (2001)
5[1]	Fethiye (Turkey) Kumluca (Turkey) Belek (Turkey) Side-Kizilot (Turkey) Anamur (Turkey) Zakynthos (Greece) Chania (Greece)	36.624/ 29.095 36.312/ 30.272 36.854/ 31.042 36.762/ 31.402 36.048/ 32.837 37.673/ 20.915 35.515/ 24.021	110 (2000) 305 (1998) 682 (2000) 270 (1998) 187 (1996) 1175 (2002) 100 (2002)
6	none		
7	none		
8 (highest level)	none		

[1] Note high number of index *C. caretta* nesting sites located in areas of considerable light pollution in 1996/1997 (category 5, yellow).
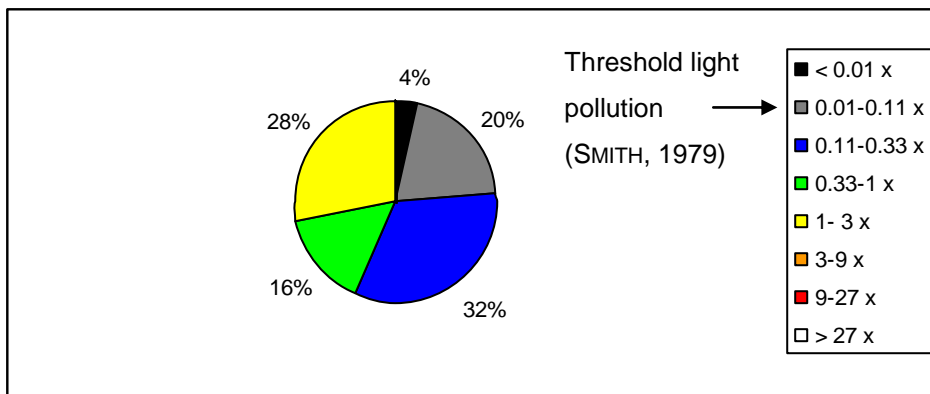
Figure 3.8: Distribution of the 25 *Caretta caretta* index nesting sites (Table 3.3) within the 8 categories of night sky brightness 1996/1997 (see Table 3.1). Note that 76% of all nesting sites compiled are located in areas which are above the threshold level of light pollution, according to SMITH (1979)

Mugla and Antalya regions, Turkey
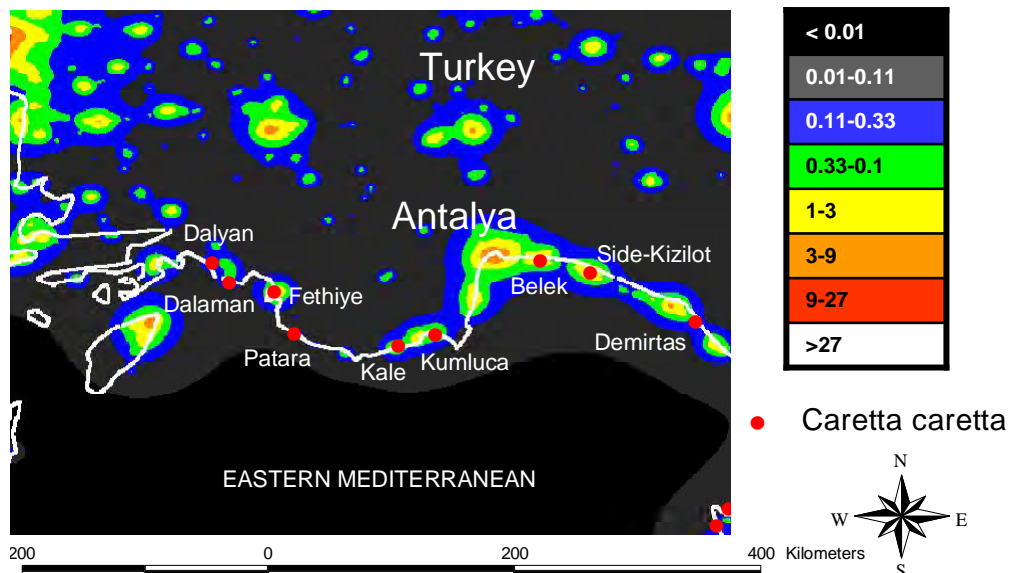
Figure 3.9: Light pollution at *C. caretta* index nesting sites in the Mugla and Antalya regions, Turkey (1996/1997). Light pollution categories according to Table 3.1. Note high light pollution levels at the top index site Belek in the Gulf of Antalya (category 5, yellow).

Cyprus; N-Cyprus; Mersin region, Turkey
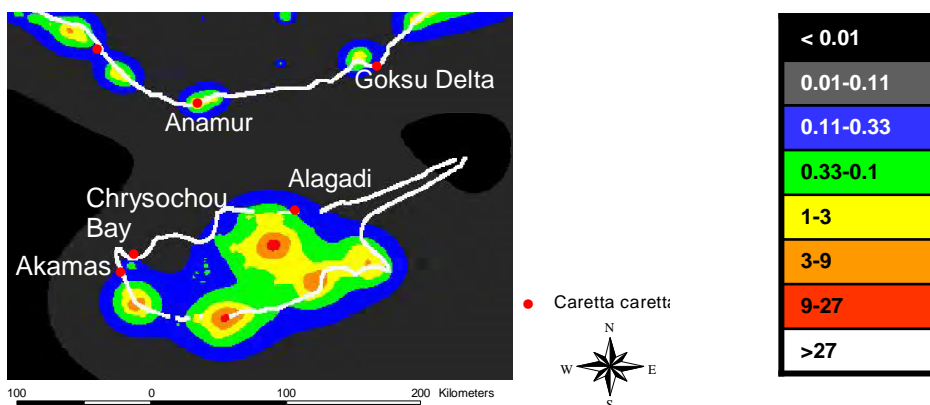
Figure 3.10: Light pollution at *C. caretta* index nesting sites in Cyprus, Northern Cyprus and the Mersin region, Turkey (1996/1997). Light pollution categories according to Table 3.1. Note comparably low light pollution (category 3, blue) at Chrysochou Bay and the Akamas Peninsula, Cyprus.

Peloponnesos, Greece
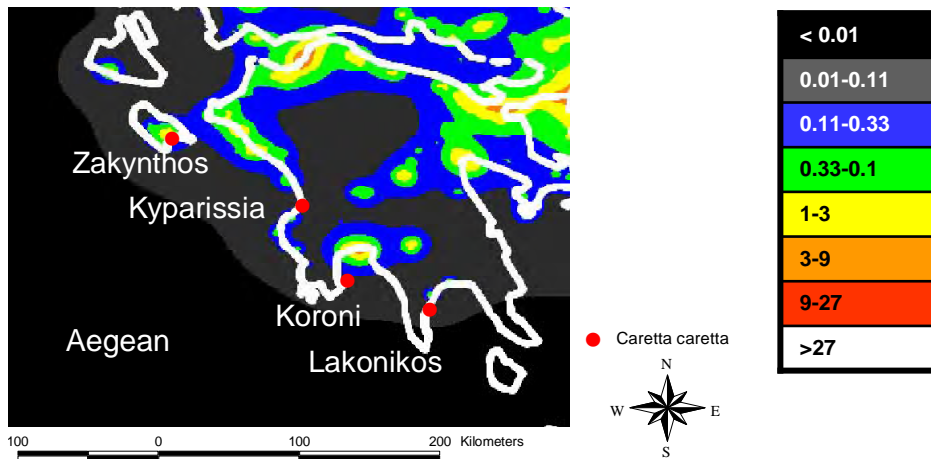
Figure 3.11: Light pollution at *C. caretta* index nesting sites in the Peloponnesos, Greece (1996/1997). Light pollution categories according to Table 3.1. Note high level light pollution (category 5, yellow) at the primary index site in Zakynthos, Greece.

Crete, Greece
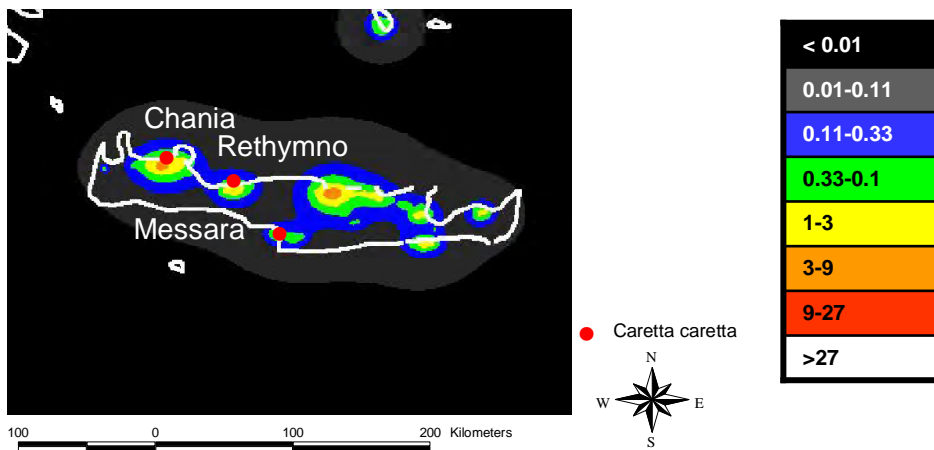
Figure 3.12: Light pollution at *C. caretta* index nesting sites in Crete, Greece (1996/1997). Light pollution categories according to Table 3.1. Note high light pollution levels (category 4, green and 5, yellow) at the three major index sites at Chania, Rethymno and Messara.

central and eastern coast of Libya
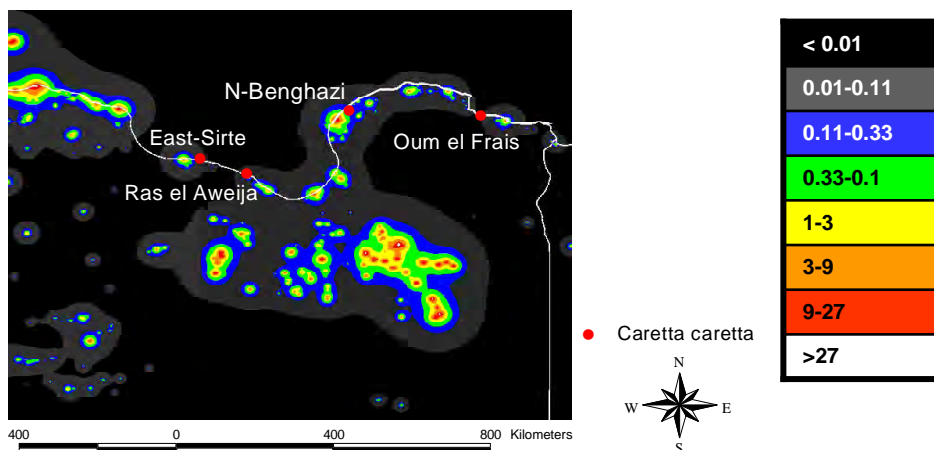
Figure 3.13: Light pollution at *C. caretta* nesting sites in Libya, North Africa (1996/1997). Light pollution categories according to Table 3.1. Note relatively low degree of light pollution (category 1, black, to 3, blue) at the four largest sites in Libya (according to LAURENT et al., 1997).

Green Turtle (*Chelonia mydas*)

Superimposition of *C. caretta* index nesting sites with light pollution maps revealed that 79% of 24 nesting sites compiled are above the threshold level of light pollution, according to SMITH (1979) (> category 2, see Figure 3.14).

The outcomes of the SPEARMAN's rank correlation test reveal that there is no significant correlation between high light pollution levels and low nest numbers for 21 *C. mydas* index sites assessed [H_1 with p (two-tailed) = 0.897697].

Table 3.4: Mediterranean *C. mydas* nesting sites and exposure to light pollution (1996/1997)

Light pollution category	Nesting site	Latitude/longitude (decimal degrees)	Nest number (year)
1 (lowest level)	North Karpaz (Northern Cyprus) South Karpaz (Northern Cyprus)	35.661/ 34.482 35.628/ 34.521	179 (2000) 34 (2000)
2	Agyatan, Yumurtalik (Turkey) Yelkoma, Yumurtalik (Turkey) Akamas Peninsula (Cyprus) Patara (Turkey)	36.587/ 35.498 36.587/ 35.498 34.948/ 32.306 36.317/ 29.245	4 (1996) 2 (1996) 75 (1990) 2 (2000)
3	Dalyan (Turkey) Tuzla (Turkey) Göksu Delta (Turkey) Akyatan (Turkey) Sogözü, Yumurtalik (Turkey) Lara/Toxeftra (Cyprus) Alagadi (Northern Cyprus) West coast of Northern Cyprus	36.834/ 28.645 36.729/ 34.903 36.297/ 34.032 36.556/ 35.319 36.776/ 35.799 34.916/ 32.323 35.334/ 33.489 35.340/ 32.935	No data 8 (1996) 12 (1998) 735 (1998) 213 (2004) 50 (1995) 68 (2002) 85 (2000)
4	none		
5 [1]	Kazanli (Turkey) Kumluca (Turkey) Belek (Turkey) Kizilot (Turkey) Samandag (Turkey) Alata (Turkey) Episkopi Bay (Cyprus)	36.809/ 34.755 36.312/ 30.272 36.854/ 31.042 36.762/ 31.402 36.061/ 35.945 36.617/ 34.332 34.670/ 32.854	128 (1996) 7 (1994) 8 (2000) 1 (1998) 21 (1999) 121 (2003) 2 (2001)
6	Alanya (Turkey) Ayia Napa (Cyprus) Latakia (Syria)	36.546/ 31.998 34.984/ 33.986 35.481/ 35.831	no data no data 104 (2004)
7	none		
8 (highest level)	none		

[1] Note high number of index nesting sites located in areas of considerable light pollution in 1996/1997 (category 5, yellow). Index nesting sites on the Karpaz Peninsula, Northern Cyprus, were the only sites completely free of light pollution in 1996/1997.
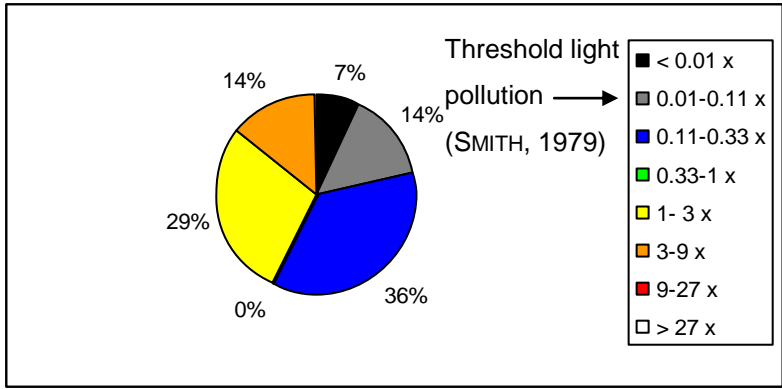
Figure 3.14: Distribution of the 24 *Chelonia mydas* index nesting sites (Table 3.4) within the 8 categories of night sky brightness 1996/1997 (see Table 3.1). Note that 79% of all nesting sites compiled are located in areas which are above the threshold level of light pollution, according to SMITH (1979).

Cyprus; N-Cyprus,Turkey; Syria
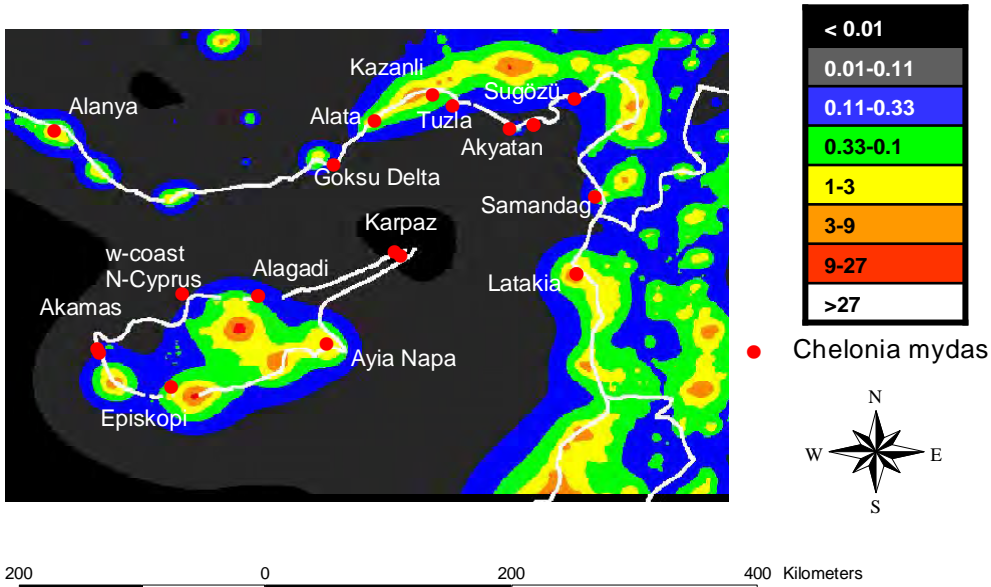
Figure 3.15: Light pollution at major *C. mydas* index nesting sites in Turkey, Cyprus and Syria (1996/1997). Light pollution categories according to Table 3.1. Note that the Karpaz-Peninsula (N-Cyprus) was the only nesting area in 1996/1997 which was free of light pollution (category 1, black).

Mugla and Antalya regions, Turkey
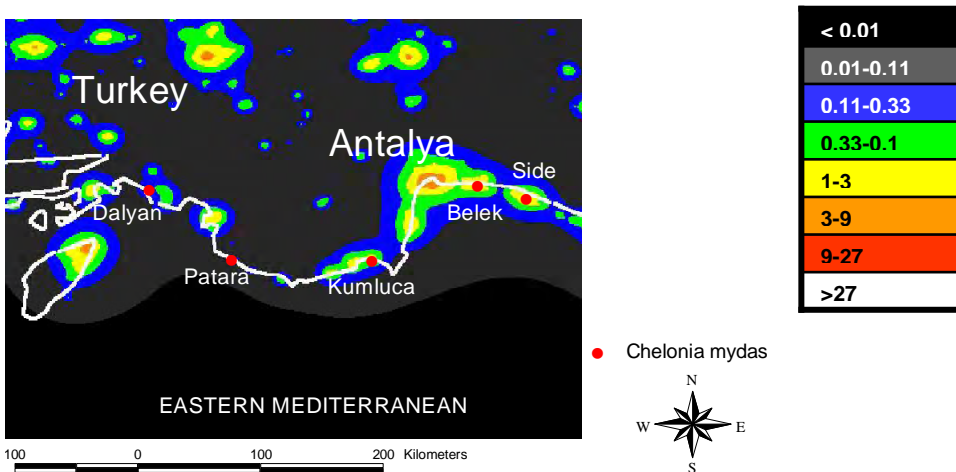
Figure 3.16: Light pollution at minor *C. mydas* index nesting sites in the Mugla and Antalya regions, Turkey (1996/1997). Light pollution categories according to Table 3.1. Note high light pollution levels in the Gulf of Antalya.

3.3 Discussion

The light pollution maps used in my thesis are adequate for the evaluation of brightness levels at sea turtle nesting sites for two decisive reasons. First, they show the artificial light propagation at sea level (CINZANO et al., 2001a). Second, the maps collect the light pollution of those sources that have a negative effect on sea turtles (Table 1.3, Figures 3.2 a, b). Hence, they are adequate for quantifying and illustrating light pollution at sea turtle nesting beaches area-wide for comparison. However these maps lack accuracy on a very small scale (< 2.8 km). Therefore they are not suitable for assessing the distribution and differentiation of artificial light sources at a specific nesting beach. In this context, my results are understood as a comparative tool for identifying light pollution "hot spots" at nesting sites, which was the primary goal in this chapter. Further analysis in the field was needed to verify these results in a case study. As a consequence, I investigated the situation in Belek, which is one of the largest *C. caretta* index sites in the Mediterranean but also known to be a negative example of coastal development and light pollution due to mass tourism. This will be discussed in Chapter 4.

Increasing light pollution is generally correlated with urbanization. Highest light pollution is found in the urban central zones of developed countries (CINZANO et al., 2001a). Data from the United Nations World Population Prospects (Tables 3.2 a, b) indicate that a number of the Top ten sea turtle nesting sites compiled in this thesis are located in countries of high population density (PD), which is human population per square kilometre. Whereas the world average PD was 45 in 2005, the PD in coastal zones is estimated to be about three times higher than the world average (SMALL and NICHOLLS, 2003). This highlights a particular risk to nesting sea turtles and their hatchlings, since they are dependent on undeveloped nesting areas. Coastal development is evident for Japan, which had a PD of 332 in 1995 and 338 in 2005, tending upwards. Due to its highly developed and urbanized status, Japan is also strongly affected by light pollution (Figure 3.6). At present, Japan holds important *C. caretta* index nesting sites on its southern beaches at the Yakushima Islands, which tend to be stable (see Chapter 2). To what extent human population growth and increasing light pollution will affect these sea turtle populations in the future should be investigated over time. Besides Japan, the situation in the United States of America is pointed out here. Whereas the average PD in the US is comparatively low (PD in 2005: 31), it is far above average in Florida (PD in 2000:

185), which holds one of the largest *C. caretta* nesting agglomerations worldwide. Florida is also seriously affected by light pollution at its nesting beaches (SALMON et al., 1995a, SALMON et al., 1995b, SALMON and WITHERINGTON, 1995, SALMON, 2003). The development of appropriate mitigation measures, such as the change of polychromatic light sources (see Chapter 1), has already been carried out at some index beaches here to reduce hatchling mortality (WITHERINGTON and MARTIN, 1996, Florida Power and Light Company, 2002). In contrast, mitigation measures for light pollution are still lacking for the Mediterranean (DEMETROPOULOS, 2001, VENIZELOS, CANBOLAT, pers. comm.). My results show that most index nesting sites in the Mediterranean are located in developed countries of increasing population density, which as a secondary effect will probably further increase light pollution. Greece and Turkey, which contribute to the global Top ten *C. caretta* nesting sites, still show high light pollution (Table 3.2 a). A particular problem for the Mediterranean coastline is mass tourism, which is evident for Greece (including Crete), Turkey, and at a progressive rate also for Cyprus. As a consequence, coastal development including light pollution causes considerable sea turtle nesting space loss, namely in Zakynthos, Greece (MARGARITOULIS, 1990, VENIZELOS, 2001) or Belek, Turkey (SAK and BARAN, 2001, CANBOLAT, 2001, AUREGGI, 2003). The example of Zakynthos is highlighted here. Though this primary *C. caretta* nesting area has protected status, there is continuous non-compliance of existing agreements. This stresses the need for tightened national and European Union-conducted inspection measures here (VENIZELOS, 2001). The situation on the Libyan coast is also highlighted, as it holds important *C. caretta* stocks which were just recently described in the literature (LAURENT et al., 1995, 1997, 1999). In 2005, Libya had a high PD (100) on its coastline. Remarkably, the four major *C. caretta* nesting areas at East Sirte, Ras el Aweija, North Benghazi and Oum el Frais (LAURENT et al., 1997) are located in areas of relatively low-level urbanization and light pollution (Table 3.3, Figure 3.13). This may indicate the preferences which females have for darker nesting areas (WITHERINGTON, 1992a). Though Libya does not yet provide long-term census data, it has an important nesting potential due to its widespread sandy beaches (LAURENT, 1995). It could be of future importance for this species, considering the nesting space lost at other beaches. Therefore its nesting habitats should be monitored with special attention. To date, Libya still has the chance to conserve its sea turtle populations by prohibiting intensive coastal development and light pollution at index nesting sites.

The outcomes of my investigation reveal that 76% of the major *C. caretta* Mediterranean index nesting sites are located in areas which were affected by light pollution in 1996/1997. Twenty-eight percent of these sites were affected by light pollution that was up to three times higher than natural ambient light in 1996/1997. As will be discussed in detail in Chapter 4, I found consistency with this assessment in the egg-laying zone of Belek. In developed areas there was an increase in Illuminance measured on the ground, which was many times higher than natural Illuminance levels at night. For *C. mydas* nesting sites I found a similar negative trend, with 79% of these index sites being above the threshold level for light pollution in 1996/1997. Twenty-nine percent of these sites were affected by light pollution that was up to three times higher, and 14% by light pollution that was even up to nine times higher than natural ambient light at night. Considering that the data on light pollution used in this chapter were obtained in 1996/1997, it is suggested here that increasing coastal development, especially in Greece and Turkey, makes the present-day situation even worse. Newer comparable light pollution maps for confirmation are not available to date, but there is an average annual increase of 5 to 10% in installed light flux in areas where no countermeasures have been taken (FALCHI, pers. comm.). Consequently a minimum twofold increase in light pollution within the last 10 years must be assumed for zones affected by coastal development and mass tourism. As light pollution is not only restricted to its source but also propagates to other regions or even countries, this highlights the urgent need for joint conservation measures for sea turtles in the Mediterranean.

The results of the SPEARMAN's tests for the Mediterranean nesting sites indicate that there was no significant correlation between increased light pollution and low nest numbers in recent years (1990-2004). High nesting was also found in areas which were seriously affected by light pollution, namely Zakynthos in Greece, or Belek, Kazanli and Alata in Turkey. One explanation for this is that these sites traditionally provide better nesting conditions, like sand structure or incubation temperature, which may compensate for increased light pollution. On the other hand, there is just as little evidence that nest numbers at illuminated sites were constant in the Mediterranean over the last years. As pointed out before, nest numbers in the Mediterranean are subject to strong annual fluctuations (Figures 2.1 a, b), which makes the assessment of the impact of light pollution difficult. But it is suggested here that nesting efforts were probably higher in the past, when there was less

coastal development and light pollution at these sites. For the Mediterranean, this will be difficult to prove due to a lack of long-term monitoring data for many sites. Moreover, a change of assessors, changing methods of sea turtle monitoring, and variable beach sizes assessed at the same nesting site may result in an inconsistency of monitoring data (see Chapter 2). This is also applicable to Belek, which was identified as an important *C. caretta* nesting site with 240 nests counted in 1979, and has provided variable nest numbers since then (1988: 226 nests, 1994: 68, 1995: 150, 1996: 153, 1997: 168, 1998: 395, 1999: 612, 2000: 682 nests) (SAK and BARAN, 2001, CANBOLAT, 2001, AUREGGI, 2003). The results of the SPEARMAN's tests may also correspond to the theory that sea turtles show high nesting site fidelity (BOWEN et al., 1992, BRODERICK et al., 2007) and consequently may not be able to shift to remote nesting sites, even if light pollution in a particular area increases. But on a smaller spatial scale I found that sea turtles avoided beach sections of high Illuminance (Chapter 4). In fact, preferred nesting sites are also ones where coastal development and light pollution is sparse (SALMON, 2003). At locations in Florida where low-level light pollution was present, nesting occurred but in lower numbers (SALMON et al., 1995a). The authors conclude that the repellent effect of light pollution is dose-dependent. As light pollution increases, more nests will be concentrated in the remaining dark areas. This spatial concentration on beaches bears higher risks for sea turtles by attracting predators and increasing hatchling mortality (see Chapter 4). At dense nesting sites there is also the risk of nest destruction by other females or increased egg mortality by bacterial infection (MILTON and LUTZ, 2003). Thus, though I could not show for the Mediterranean that in areas of increased light pollution sea turtles nest numbers are generally reduced (SPEARMAN's tests), it is strongly suggested that artificial lights set close to the shore do affect the nest-site selection of females here. This is supported by data collected in the USA (WITHERINGTON, 1992a, SALMON et al., 1995a). The negative effect which artificial light has on hatchlings was also confirmed in many studies in the United States (MCFARLANE, 1963; PHILIBOSIAN, 1976; WITHERINGTON, 1992b; WITHERINGTON and MARTIN, 1996). Statistical comparison of future nesting data based on constant monitoring methods, and the simultaneous monitoring of light pollution over time is needed to verify my results. This could be a starting point for further research on this issue. Based on the alarming outcomes of the presence of light pollution at Mediterranean nesting sites (Figures 3.8, 3.14) I investigated this problem in the field.

4 The *Caretta caretta* index nesting site Belek, Turkey: A case study

4.0 Introduction

According to the Marine Turtle Specialist Group, coastal development and light pollution are major threats to the survival of sea turtles[16]. This chapter will focus on this problem, investigating the situation of Belek on a small scale (ground analysis). The Mediterranean, with a minimum 21 *C. caretta* index nesting beaches (Table 3.3), contributes to the Top ten nesting sites in the world. It is also important for a local *C. mydas* subpopulation (see Chapter 2). Belek, which is located 30 km east of Antalya on the so-called Turkish Riviera (Figure 4.1), has the largest *C. caretta* nesting agglomeration in Turkey and is probably the second largest site in the entire Mediterranean, with 682 nests counted in 2000 (CANBOLAT, 2001). Like a number of other sites in the Mediterranean, Belek faces the threats of mass tourism and coastal development, which are affecting sea turtles here (DEMETROPOULOS, 2001). Satellite data revealed that Belek is located within an area of considerable light pollution (see Chapter 3).
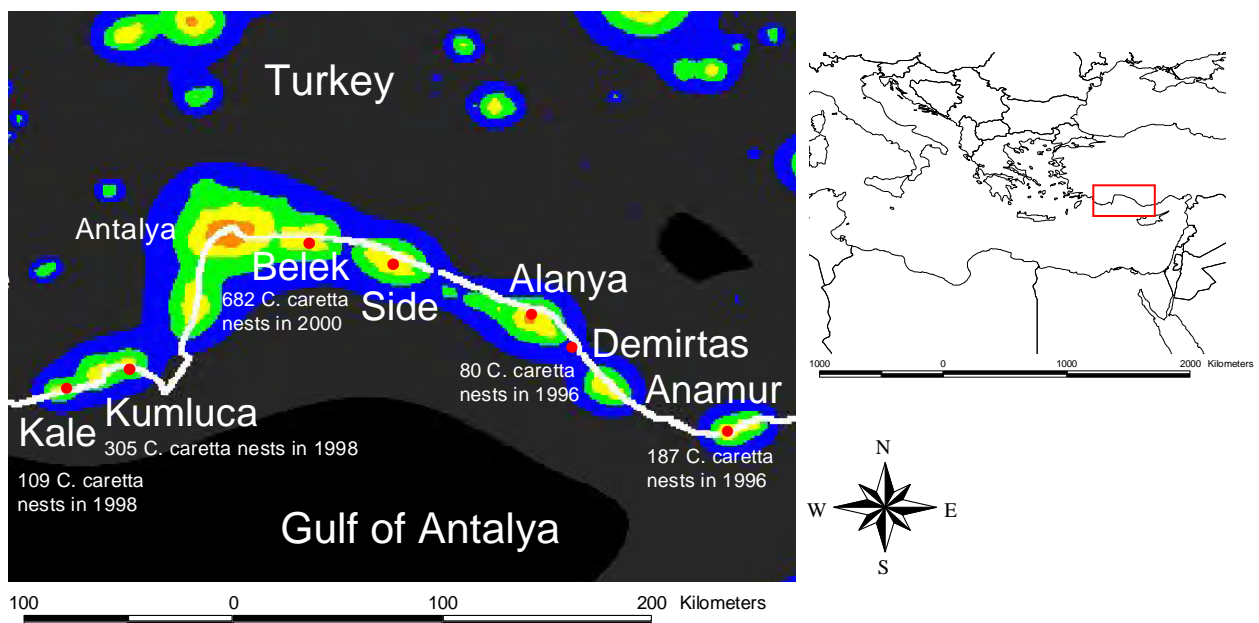
Figure 4.1: Location of Belek, Turkey, on the Gulf of Antalya, Eastern Mediterranean. This site holds one of the largest *C. caretta* agglomerations in the Mediterranean, but also had high-degree light pollution with an up to 3-fold increase in ambient light at night in 1996/1997 (see Table 3.1).

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[16] www.iucn-mtsg.org/

In 1984, a beach stretch of 11.3 km in Belek was declared a Tourism Centre by a governmental decree, followed by the implementation of the Belek Tourism Investors Union (BETUYAB) in 1989. Since then BETUYAB has promoted mass tourism in this region, evidenced by the construction of recreational facilities and the increasing number of tourists, mainly from Russia and Germany (Figure 4.2). The tourist peaks overlap with the arrival of the nesting females and hatchling emergence in the summer months (CANBOLAT and NALBANTOGLU, 2001).
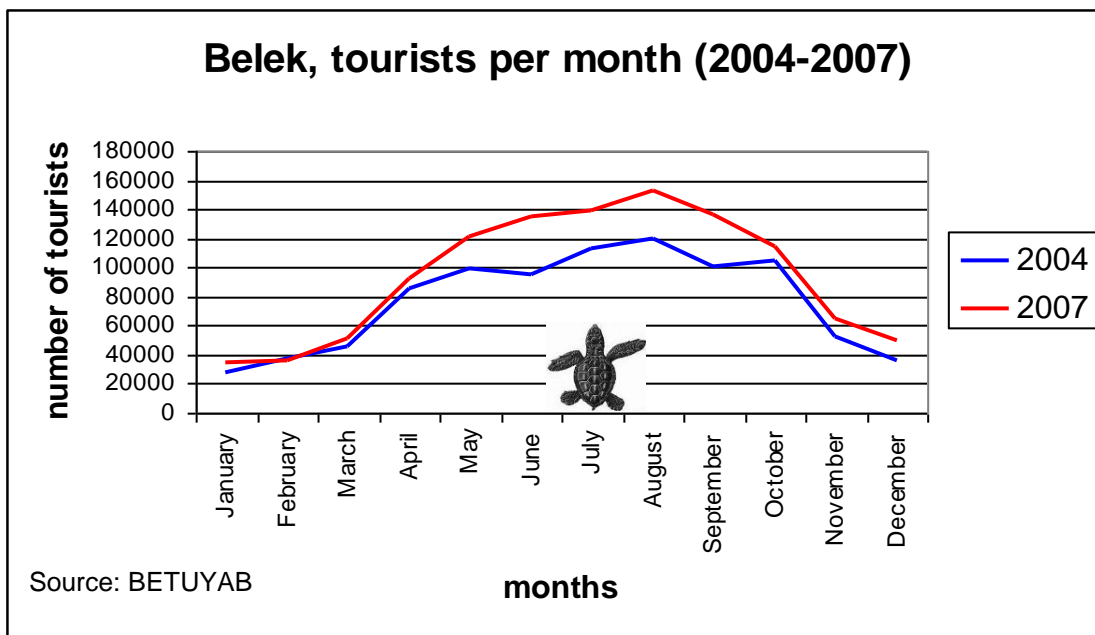
Figure 4.2: Belek tourism area. Increase in the guest numbers per month from 2004 to 2007. The peaks in June, July and August overlap with *C. caretta* nesting and hatchling emergence in this area.

BETUYAB advertises its efforts to encourage conservation of the environment and of nature, and is also aiming to be an international model for sea turtle conservation. For these efforts, the Republic's Ministry of the Environment official research project tender was awarded to BETUYAB in 2000. BETUYAB has also won awards in several international environment and tourism competitions, which has also been recognized by the United Nations Department of Economic and Social Affairs. However, on its web page[17], BETUYAB also highlights its primary goal of increasing the quality of tourism, which involves further development of infrastructure such as the construction of coastal roads and the illumination of roads and tourist facilities. Belek is already known to be a negative example for intensive coastal development due to mass tourism (SAK and BARAN, 2001, CANBOLAT and NALBANTOGLU, 2001,

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[17] www.betuyab.org

AUREGGI, 2003). This includes tourism facilities in close proximity to the shoreline, vehicle traffic on coastal streets and on the beach, water sports areas, and human activity in front of hotels. This leads to disturbance of females and sand compaction on the beach, which affects the incubation of the eggs. The general problem of light pollution (see Chapter 1) has been diagnosed for Belek. SAK and BARAN (2001) suggested that light pollution caused high hatchling mortality in the 1996 and 1997 nesting seasons. According to these authors, a minimum of 781 hatchlings died due to light pollution in 1996, whereas 8117 hatchlings managed to reach the sea. Hatchling mortality increased to 1091 in the 1997 nesting season, and only 7082 hatchlings managed to reach the sea in the same year. The authors further conclude that additional 1263 hatchlings, which were counted as "lost on the beach" in both years, were in fact disoriented by artificial lights on the beach and probably perished further landwards. CANBOLAT and NALBANTOGLU (2001) conducted research to identify areas where artificial lights affected hatchlings. For this, the "Artificial Light Diagram" was used, which allowed an assessment of the crawling direction of hatchlings and the evaluation of disorientation rates. Investigation in the Belek nesting area revealed that the overall ratio of hatchlings that failed to orient seawards was 33.5 % in 1999 and 2000 (CANBOLAT and NALBANTOGLU, 2001). These research efforts were important for highlighting the problem of light pollution in Belek, but as yet there is no study available specifying the increase in coastal development and light pollution in Belek over time for comparison. My study focuses on the major sources of light pollution, quantifying and specifying those artificial lights on a small spatial scale that are interacting with the females and hatchlings. This is the first step in developing effective mitigation measures, which are still lacking at this important nesting site. Another concomitant of coastal development in Belek is noise pollution (see Chapter 1). It is assumed that noise has a negative effect on sea turtle females on nesting beaches (MARGARITOULIS, 1990, CANBOLAT and NALBANTOGLU, 2001). This is supported by preliminary data from the *C. caretta* index nesting site in Zakynthos, Greece (National Aviation Service, 1990). But so far the use of a scientific approach to investigate the impact of noise on sea turtles on land has been lacking. A second objective of this chapter is a collection of data on noise pollution, including the identification of its sources in the egg-laying zone of Belek. This study aimed to provide a background for investigating the possible effects of acoustic noise on sea turtle hatchlings (see Chapter 5), which is completely unknown to date.

Spatial characteristics of the study area

The *C. caretta* nesting area Belek has a total beach length of 29.3 km (Table 4.1) which is divided into four sections based on different geographical features and beach usage (CANBOLAT and NALBANTOGLU, 2001; see Appendix 3, Tables 4.3 a, b). I conducted research on the two western Sections I and II, which were monitored by Dr. CANBOLAT from the Hacettepe University, Ankara, and his team in the nesting season 2005. Section I has a beach length of 4.7 km and is located between the Aksu River mouth in the west and the hotel "Asteria" in Hotel zone 1 in the east. A second river, the Besgöz, runs parallel to this section's central shoreline, having its estuary 1.7 km east of the Aksu River mouth. Section II has a total beach length of 11.3 km and is located between Section I in the west and the Acisu River mouth in the east. This section is a designated tourism development area, which is promoted by BETUYAB (Figure 4.3).

Table 4.1: Division of the Belek nesting area (according to CANBOLAT and NALBANTOGLU, 2001)

Division	Description	Location	Beach length (km)
Section I: (monitoring area "AY" in 2005)	Natural Site Area	Aksu River - Hotel Asteria	4.7
Section II: (monitoring areas "AOY" and "OY" in 2005)	Tourism Development Area	Hotel Asteria - Acisu River	11.3
Section III:	Specially Protected Area	Acisu River – Koprucay	6.7
Section IV:	Specially Protected Area	Koprucay – Sarisu Stream	6.6
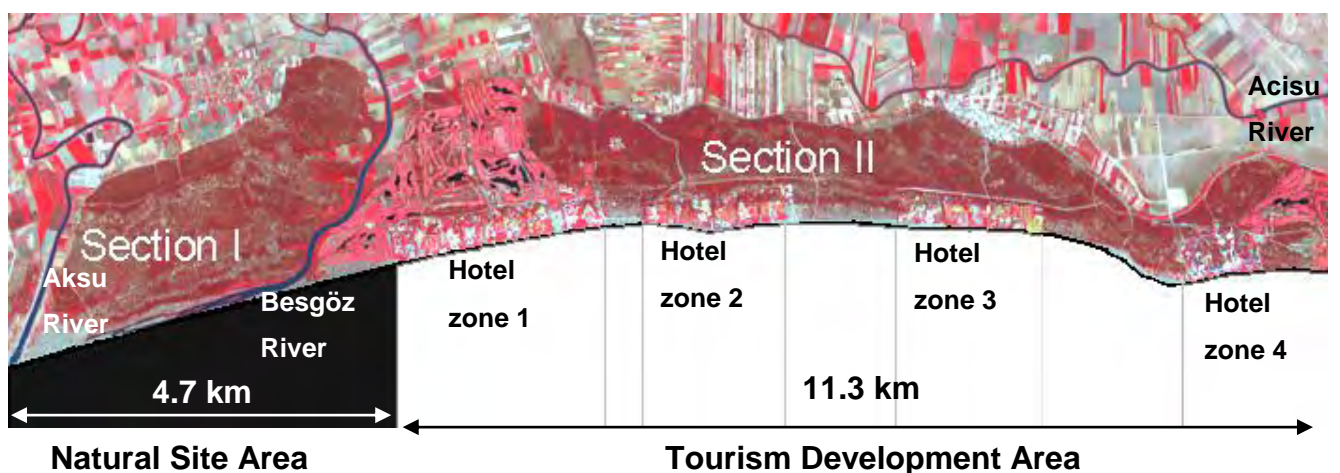
Figure 4.3: *C. caretta* nesting area Belek. Section I (Natural Site Area) and Section II (Tourism Development Area). Areas between the hotel zones were not yet covered with buildings in September 2005. Sections III and IV (not covered in this thesis) are located east of Section II.

4.1 Methods

An initial approach of my study in Belek was to assess the general degree of coastal development in the egg-laying zone and its progression over time. For this, I evaluated ASTER daytime satellite data, showing the progression of coastal development in the Belek nesting area from 2000 to 2004. Coastal development was also investigated in the field in 2005. A focus was made on quantifying light pollution here. The number of major stationary artificial light sources per kilometre was counted over 16.1 km total beach length and defined as Artificial Light Source Density (ALSD). The models of light sources (lamps) were specified. At representative locations Illuminance levels were measured. Technical parameters of light pollution assessed in the field were correlated with female nesting density, hatchling disorientation, and hatchling mortality in the nesting area. Basic research was conducted on noise pollution in the study area. High intensity sound sources were recorded in the field and evaluated with regards to sea turtle acoustic perception ability. Moreover, field experiments were conducted with *C. caretta* hatchlings to test the combined impact of artificial lights and high intensity sounds on their orientation behaviour.

4.1.1 Progression of coastal development in Belek from 2000-2004

ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer) data of the TERRA satellite platform, which was launched in 1999, was evaluated. The satellite obtains high-resolution (15 to 90 square metres per pixel) images of the Earth in 15 bands, scanning 14 different wavelengths of the electromagnetic spectrum, ranging from visible to thermal infrared light. Bands 1 to 4 of the ASTER lens scan the visible/near infrared (VNIR) spectrum (Table 4.2). The high-resolution properties of these VNIR bands are adequate to discern infrastructure (hotel zones, coastal roads, construction sites) on the ground. The 3N band (Nadir) and the 3b band scan in the same spectrum and enable stereo 3D-projections of the landscape. Bands 5 to 10 scan in the shortwave infrared (SWIR, 1600-2430 nm) bands 11 to 15 in the thermal infrared (TIR, 8125-11650 nm) spectrum. SWIR and TIR bands were not evaluated for the purpose of this thesis.

Table 4.2: Visible near-infrared bands of the ASTER lens

Band numbering	Band specification[1]	Wavelength	Resolution
B1	VNIR Band 1	520-600 nm	15m
B2	VNIR Band 2	630-690 nm	15 m
B3	VNIR Band 3N	760-860 nm	15 m (Nadir) vertical projection
B4	VNIR Band 3B	760-860 nm	15 m backward scan: 27.6º

[1] VNIR = visible and near-infrared light

The progression in coastal development from July 2000 to July 2004 was assessed. For this, ASTER data taken on 13th July 2000 and 24th July 2004, covering the Belek nesting area, were used. These data were kindly provided Dr. Hossein YAMOUT[18]. High-resolution satellite data are also commercially available on the ASTER web page[19].

Data was evaluated in ENVI 4.2 (Environment for Visualizing Images) image-processing software. For the visible/near-infrared bands 1-3 the colour of the images was set in the RGB (Red-Green-Blue) modus. Projection according to R=3N, G=2 and B=1 was chosen for visualisation of satellite data (Figures 4.6 a, b). This false colour composite projection is commonly used in remote sensing for highlighting vegetation in red, as it provides a good contrast to developed areas, which appear white or grey due to high reflection. The red colour for vegetation is chosen because vegetation reflects a lot of near-infrared light. This spectrum, though being invisible to the human eye, is often associated with red, hence facilitating better visualisation in maps. Sediment-laden or shallow water (coastal zone) in this projection appears blue, whereas clear, deep water is dark blue. Urban areas look blue-grey to white. The ASTER satellite data was also used for creating high-resolution maps which were applicable for illustrating the location of stationary artificial light sources and sea turtle nests in Belek (Appendix 3).

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[18] Institute of Geography, University of Bonn

[19] http://asterweb.jpl.nasa.gov/

4.1.2 Coastal development and light pollution in Belek (2005)

In September 2005, the coastal development in Belek was inspected, covering 16.1 km total beach length in Sections I and II. This included coastal streets, hotels and construction sites. Overall light pollution was photographed at night between 11 pm and 5 am. For this, a Nikon F-80 autofocus camera with a Zenitar M" 8/16 mm fisheye lens and Fuji Sensia 400 ASA 135-36 film was used. Stationary light sources that contributed to high light pollution levels were captured separately at night by using a Tamron AF 28 200 mm (IF) lens. The technical parameters of these lamps, including model, wattage, mounting and distance to shoreline, were recorded. For this, pictures were taken during daylight, using the Tamron AF lens with a Hoya Skylight (1B) UV- filter and Fuji Velvia 50 ASA films. All pictures were digitalized using a Nikon LS-5000 scanner. The images were saved on a computer and processed with Photoshop CS for printout. The coordinates of stationary artificial light sources in the field were obtained with a portable Garmin etrex GPS tool (accuracy < 15 m). The results are given in Chapter 4.2.

For statistical evaluation of my results, Sections I and II were subdivided in my thesis, based on different spatial characteristics of the beach.

Table 4.3 a: Subdivision of Section I, Belek, see Appendix 3 a, b

Section I (Natural Site Area)			Total length 4.8 km
Subdivision[1]	description	location	beach length (km)
West	Natural Site Area	Aksu River mouth - Besgöz River mouth	1.66
Centre	Natural Site Area	Besgöz River mouth - Besgöz northward bend	2.15
East	Natural Site Area	Besgöz northwards bend - Hotel Asteria (Section II)	0.98

[1] Area "West" is characterized by low-degree coastal development between the Besgöz and Aksu River mouths. Area "Centre" is characterized by its flat beach profile lacking higher dunes along the Besgöz River (see Appendix 3 a). Area "East" is characterized by smaller dunes and anthropogenic vegetation (golf areas) in its hinterland.

Table 4.3 b: Subdivision of Section II, Belek (from west to east, see Appendix 3 c-f)

Section II (Tourism Development Area)			Total length 11.3 km
Subdivision[1]	description	location (hotels)	beach length (km)
Hotel zone 1	Tourism Development Area	Asteria, Megasaray, Adora, Tat Beach, IC, Magic Life Sirene, Sirene Golf, Kempinski, Kaya	2.57
Public area 1	Tourism Development Area	Beach area between hotels Kaya and Justiniano	0.49
Hotel zone 2	Tourism Development Area	Hotel Justiniano, Sun Zeynep, Club Justiniano, Altis, Papillon Belvil, Marmara Bellis, Attelia	1.74
Public area 2	Tourism Development Area	Beach area between hotels Attelia and Magic Life	1.34
Hotel zone 3	Tourism Development Area	Magic Life, Belconti, Arcadia, Grida, Ali Bey	1.76
Public area 3	Tourism Development Area	Beach area between hotels Ali Bey and Sillyum	1.84
Hotel zone 4	Tourism Development Area	Sillyum, Cornelia, Pine Beach, Atlantis, Gloria Verde, Zeugma, Rixos Premium, Cesars Temple	1.56

[1] Hotel zones are agglomerations of constructions and buildings that are located within 100 m of the shoreline. Public areas are beaches that were located between the hotel zones and were not covered with buildings in September 2005.

Light pollution

An overall assessment of light pollution was made by visual inspection and nighttime photography in the field. Light pollution was differentiated based on its location and visibility. For this, the terms "sky glow" and "direct irradiation" are used constantly in my thesis. Sky glow is artificial light reflected from illuminated objects and refracted in the surrounding atmosphere. The light source causing sky glow may not be visible from the beach. In contrast, direct irradiation is caused by a light source that is clearly visible from the beach. Direct irradiation measurement was based on the technical parameters of the light source (lamp), including model, wattage, mounting and emitted wavelength (see Chapter 1, Table 1.3).

Artificial Light Source Density (ALSD)

The term ALSD was introduced in this work to quantify artificial light sources on the beach. The ALSD is defined as the number of major stationary artificial light sources per one kilometer beach length. These are lamps, such as High Pressure Sodium Vapour (HPS) and Metal Halide (MH) lights, which contributed to high Illuminance and could be clearly identified technically in the field. The ALSD was used for statistical analysis (SPEARMAN's rank correlation test) to calculate the effects of light pollution on female nesting density and hatchling disorientation (Chapter 4.2.1).

Illuminance

To quantify light pollution on the nesting beach on a small scale the illuminance was measured. Illuminance measurement is a standard method in photometry to evaluate the efficacy of lighting installations. Illuminance (symbol E_V) is defined as the total luminous flux incident on a surface per unit area, according to Equation 4.1:

E_V = luminous flux / unit area

luminous flux (SI unit: lumen, lm)
unit area (in m^2)

The illuminance depends on the luminous intensity (I_V) of the light source (lamp) and the distance to it. The resulting illuminance attenuates with $1/r^4$ (r = distance to the light source in metres).

Table 4.4: SI photometry units[20]:

Quantity	Symbol	SI unit	Definition
Luminous flux	F	Lumen (lm)	Perceived power of light
Luminous intensity	I_V	Candela (cd)	Measure of wavelength weighted (555 nm) power emitted by a light source
Luminance	L_V	Candela per square metre (cd/m^2)	Density of luminous intensity in given direction
Illuminance[1]	E_V	Lux (lx)	Light incidence on a surface

[1] Conversions: 1 lux = 1 lm/m^2 = 1 sr x cd/m^2. Steradian (sr) is the SI unit of solid angle.

Illuminance was measured in the field in the egg-laying zone (within 25 m of the shore) at locations which represented average light pollution levels in the given area. For this, a calibrated BEHA Unitest digital Luxmeter 9342 with a silicium sensor was used. This device has a log function which was used to manually record the measured values. The results are given in Table 4.14.

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[20] The International System of Units (2008)

4.1.3 Ambient sounds and noise pollution

For differentiation in this thesis, the term "sound" relates to natural sounds (breaking waves/surf sounds), whereas the term "noise" is used for anthropogenic sounds (e.g. discotheque noise). The first step in this study was to identify prevalent sources of sound in the field. Natural sounds and noise pollution in the coastal area of Belek were measured. To avoid the problem of masking by background noise (PATTERSON and GREEN, 1978), I conducted measurements on surf sounds in areas of low coastal development (Section I). Surf sounds were measured in the early morning, when ideally no major anthropogenic sound source was present on the beach. In contrast, noise was measured in areas of high coastal development (Section II), typically in the late evening or in the night, when there were increased noise levels due to higher human activity on the beach (e.g. discotheques, vehicle noise, fireworks from hotels). However, my measurements on noise must be considered approximations, as the masking effect by surf sounds cannot be excluded, the latter being a permanent sound source in the field. To avoid errors when differentiating sound sources, repeated measurements of sound levels (in db SPL re 20 μPa) and frequencies (Hz) were made. In a second step, it was measured to what extent the ambient sound levels changed within the range of sea turtle nests. For this, measurements were made in the egg-laying zone at 5 to 25 m distance from the shoreline, in which most of the nests were found (pers. observation, see Appendix 3). The aim was to find out if an anthropogenic sound source at high intensities, set close to a sea turtle nest, would be able to mask ambient wave sounds here (see Chapter 1). In a third approach I compared my measurements with literature data on sea turtle acoustic perception ability from RIDGWAY et al. (1969) to assess if sea turtles in fact would be physically able to perceive the ambient sounds and anthropogenic noise at given distances.

Measuring techniques

Measurements were made using a Fast Fourier Transformation (FFT) Audio Frequency Analyser, model HMB-TEC A316 with integrated condenser measuring microphone (frequency response: 15 Hz – 20 KHz, output sensitivity: 5mV/Pa). In combination with a laptop model Toshiba Satellite Pro 4200, this tool allowed the measuring of frequency curves and sound levels in the field. Unweighted Sound Pressure Levels in dB (SPL) were recorded using the HMB-TEC A316 integrated sound level meter. This was necessary with regard to differences in sea turtle sound

perception capabilities compared to humans, which made the use of A, B, or C weighted SPL measurements less adequate. As outlined in the general introduction (see Chapter 1), the sound level of a sound source attenuates logarithmically with increasing distance (ZÜRCHER and FRANK, 2004).

For a <u>line source</u> like surf sound, there is a decrease of <u>3 dB SPL</u> per doubling of distance, according to Equation 4.2:

$$L_2 = L_1 - 10 \times \lg\left(\frac{r_2}{r_1}\right)$$

For a <u>point source</u> like discotheque or vehicle noise, there is a decrease of <u>6 dB SPL</u> per doubling of distance, according to Equation 4.3:

$$L_2 = L_1 - 20 \times \lg\left(\frac{r_2}{r_1}\right)$$

L_2: Sound Pressure Level in distance r_2
L_1: Sound Pressure Level in distance r_1
r_2, r_1: distance from sound source

These equations were used to assess the effect of a sound source over a distance and to verify my measurements on sound levels. The results are given in Chapter 4.2.

Sea turtle acoustic perception ability

It was calculated if sea turtles would be physically able to perceive the aerial sounds that I measured with the HMB-TEC A316 Frequency Analyser in the field. For this, I compared my measurements with electrophysiological data available for juvenile *Chelonia mydas* (RIDGWAY et al., 1969) (see Figure 1.2). Comparable data testing aerial sounds on *Caretta caretta* are not available to date. For comparison of literature data, I used physical conversions. This was necessary as the authors specified units of pressure, whereas my FFT Audio Analyser calculated output voltages (V_{out}). In a first step, auditory sensitivity (in dB re 1 dyne/cm^2) for pure tones (50 Hz to 1000 Hz) tested on *C. mydas* by RIDGWAY et al. (1969) were converted to dB (SPL re 20 µPa), according to Equation 4.4:

0 dB re 1 dyne/cm^2 = 1 dyne/cm^2 = 0.1 Pa = 74 dB (SPL)

Note: Dyne and Pascal are units of pressure. The unit Pascal is established as European standard, whereas dyne is commonly used in the United States. The Sound Pressure Level in decibels is the acoustic measuring standard used in sound level meters.

Table 4.5: *C. mydas* auditory sensitivity and conversion of units of pressure

Frequency tested:	*C. mydas* auditory sensitivity[1]:	Conversion 1:	Conversion 2:
(50 Hz)	+18 dB re 1 dyne/ cm²	7.943282 dyne/ cm²	= 92 dB(SPL)
(100 Hz)	-10 dB re 1 dyne/ cm²	0.316227 dyne/ cm²	= 64 dB(SPL)
(200 Hz)	-20 dB re 1 dyne/ cm²	0.1 dyne/ cm²	= 54 dB (SPL)
(300 Hz)	-25 dB re 1 dyne/ cm²	0.056234 dyne/ cm²	= 49 dB(SPL)
(400 Hz)	-35 dB re 1 dyne/ cm²	0.017782 dyne/ cm²	= 39 dB(SPL)
(500 Hz)	-20 dB re 1 dyne/ cm²	0.1 dyne/ cm²	= 54 dB (SPL)
(600 Hz)	-18 dB re 1 dyne/ cm²	0.125892 dyne/ cm²	= 56 dB(SPL)
(700 Hz)	-10 dB re 1 dyne/ cm²	0.316227 dyne/ cm²	= 64 dB(SPL)
(800 Hz)	-8 dB re 1 dyne/ cm²	0.398107 dyne/ cm²	= 66 dB(SPL)
(900 Hz)	0 dB re 1 dyne/ cm²	1 dyne/ cm²	= 74 dB(SPL)
(1000 Hz)	+5 dB re 1 dyne/ cm²	1.778279 dyne/ cm²	= 79 dB(SPL)

[1] *C. mydas* juvenile auditory sensitivity for airborne sounds at given frequency required to produce a cochlear potential of 0.1 µV (according to RIDGWAY et al., 1969).

In a second step, the output voltage (V_{out}) of my analyser integrated condenser microphone at given dB (SPL) (Table 4.5: conversion 2) was calculated, according to Equation 4.5:

$$V_{out} = SPL_{measured} + SPL_{microphone}\ (\text{- dB Pa}) + \text{sensitivity}_{microphone}\ (\text{-dBV/Pa})$$

V_{out} = output voltage in dB V

$SPL_{measured}$ = variables from Table 4.5, in dB (SPL) (juvenile *C. mydas* hearing sensitivity)

$SPL_{microphone}$ = 94 dB SPL (1 Pa), reference point at 1KHz for specifying the sensitivity of microphones (European standard).

Sensitivity $_{microphone}$: Sensitivity of the microphone (specific value) = 5 mV/Pa.

To convert dB (SPL) to dB (V), the absolute sound pressure in dB (Pa) was calculated. This is the sound pressure level in decibels referred to 1 Pascal (Pa), according to Equation 4.6:

dB Pa = dB(SPL) + 20 × log 20 µPa

dB Pa = dB (SPL) - 94 dB

Translation from the absolute sound pressure level to a voltage is specified by the sensitivity of the microphone (specific value). Conversion for the microphone sensitivity was made for use in Equation 4.5, according to:

5 mV/Pa = - 46.0206 dBV/Pa

Table 4.6: *C. mydas* auditory sensitivity and conversion to output voltages on the HMB-TEC A316 Frequency Analyser

Frequency tested	auditory sensitivity *C. mydas* (variables from table 4.5)	SPL microphone (constant)	Sensitivity microphone (constant)	Output voltage on HMB-TEC A316[1]
(50 Hz)	92 dB(SPL)	-94 dB(SPL)	- 46 dBV/Pa =	- 48 dBV
(100 Hz)	64 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 76 dBV
(200 Hz)	54 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 86 dBV
(300 Hz)	49 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 91 dBV
(400 Hz)	39 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 101 dBV
(500 Hz)	54 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 86 dBV
(600 Hz)	56 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 84 dBV
(700 Hz)	64 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 76 dBV
(800 Hz)	66 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 74 dBV
(900 Hz)	74 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 66 dBV
(1000 Hz)	79 dB(SPL)	-94 dB(SPL)	-46 dBV/Pa =	- 61 dBV

[1] Output voltages on the FFT Audio Analyser are calculated for juvenile *C. mydas* auditory sensitivities from 50 Hz to 1 KHz (RIDGWAY et al., 1969).

The comparison of juvenile *C. mydas* hearing sensitivity and the measured wave/surf sound audio spectrum is shown in Figure 4.18.

4.1.4 Female nesting density, hatchling disorientation and mortality

For evaluating the impacts of coastal development and light pollution on females and hatchlings in this thesis, the monitoring raw data were kindly provided by Dr. Ali Fuat CANBOLAT[21]. All data on nesting density, hatchling disorientation and mortality, which are listed in Appendix 2, were obtained by Dr. CANBOLAT and his team and remains his property. I conducted fieldwork in September 2005 as a volunteer, doing morning patrols and data collection with this team. However, evaluation on nesting density, hatchling disorientation and hatchling mortality in this thesis is based on monitoring data for the complete nesting season 2005. Of about 700 *Caretta caretta* nests identified in the entire Belek nesting area in 2005 (Sections I, II, II and IV; see Table 4.1), approximately 250 nests were recorded in Sections I and II (CANBOLAT, pers. comm.). I evaluated the data available for 139 nests in Sections I and II, including nest coordinates, emergence dates, number of hatchling tracks found, number of disoriented hatchlings and hatchlings found dead on the beach. For the remaining nests, which were mainly located in Section I, I did not have data.

Methods of sea turtle monitoring in Belek

The staff of the Hacettepe University conducted dayshifts and nightshifts during the *C. caretta* nesting season 2005, which started in May and ended in September. During the peaks in female emergence from May to July, night patrols were conducted between 10 pm and 3 am in Sections I and II on the three designated beach areas, one in Section I (monitoring area "AY"), and two in Section II (monitoring areas "AOY" and "OY"), to tag and measure females coming ashore (see Chapter 2.0). The staff determined the geographical position (latitude/longitude in decimal minutes) of the nests, using a Garmin-etrex GPS tool (accuracy < 15 m). Nests identified were partly marked with wooden stakes, or protected with cages in front of hotels. In my thesis, the exact positions of nests that were determined by GPS were used to assess the nesting density, defined as nests per kilometre. This is a standard method used in sea turtle beach monitoring. Besides conducting nightshifts, daily beach patrols were done on a routine basis between 6 am and 12 pm. Nests that were not discovered during night patrols were identified by females' tracks on the nesting beach and certain changes in the sand structure where nests were laid. Day- and nightshifts were also conducted to evaluate hatchling emergence

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[21] Hacettepe University, Ankara

success after 50-60 days egg incubation period. Hatchling tracks were identified in the sand usually the day after hatchling emergence. Tracks identified were used to assess how hatchlings oriented in the field. In this work, statistics conducted on hatchling disorientation are based on these data of hatchling track identification (see Figure 4.4). After the latest hatchling emergence event in a nest, which occurred at intervals within one to four nights, nests were excavated and remaining dead or alive hatchlings counted. The sea turtle staff also counted the number of eggshells for statistical evaluation. Besides assessing hatchling disorientation, mortality rates too were examined by visual inspection of hatchling tracks in the field. The number of hatchlings per nest that were found dead on the beach was counted. Based on this data, statistical analysis was done to assess hatchling mortality in my thesis. All statistics conducted on nesting density, hatchling disorientation and mortality are based on the sea turtle monitoring data, which are summarized in Appendix 2.

Nesting density

Based on the monitoring data provided in Appendix 2, the nesting density, which is defined as number of sea turtle nests per kilometre, was calculated for Sections I and II. The positions of all nests for which GPS data (accuracy < 15 m) was available were entered in high resolution satellite maps (Appendix 3). The nesting density was compared per beach sections as defined in this thesis (Tables 4.3) and correlated with the Artificial Light Source Density (ALSD) in the given area (Figure 4.16). Statistical evaluation was made according to Table 4.7.

Table 4.7: Statistical tests to assess nesting densities in Belek, Sections I and II (2005)

Test name	Assessment[1]	H_0	H_1
Student's t-test	Are nesting densities equal in Sections I and II?	The nesting densities are <u>equal</u> in the two beach areas	The nesting densities are <u>unequal</u> in the two beach areas
SPEARMAN`s rank correlation test	Is there a negative correlation between the Artificial Light Source Density (ALSD) and the nesting density?	There is <u>no correlation</u> between the ALSD and the nesting density	There is a <u>negative correlation</u> between the ALSD and the nesting density

[1] Probabilities for the Student's t-test were assessed one-tailed, for the SPEARMAN'S test two-tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$. Statistical evaluation was made using VassarStats statistical computation software.

Hatchling disorientation

CANBOLAT (pers. comm.) defines hatchling disorientation for an individual as showing an initial heading direction opposite to the sea. This definition was used to assess disorientation rates in Belek 2005, and is used in this chapter to keep data consistency. Assessment on hatchling disorientation was made based on the crawl tracks of hatchlings in a virtual circle of 4 m in diameter set around a sea turtle nest in its centre, according to Figure 4.4. This method allows correct assessments of hatchling orientation trends (SALMON, 2003, SALMON and WITHERINGTON, 1995).
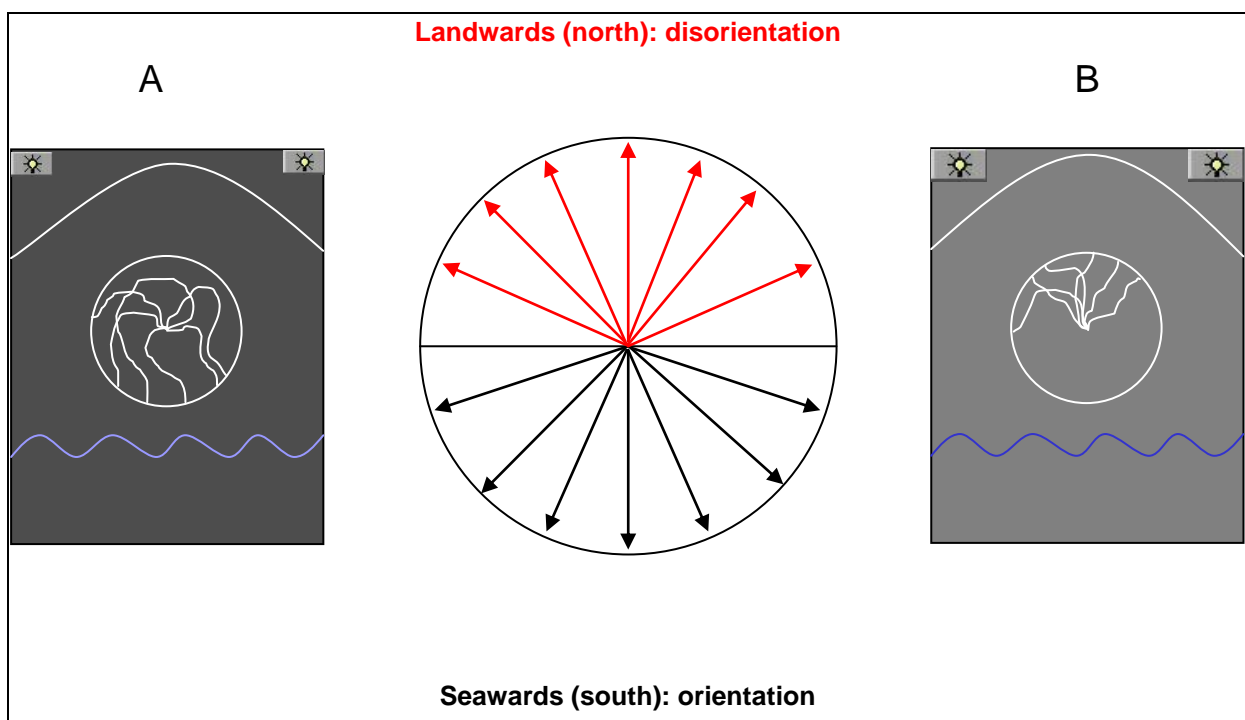
Figure 4.4: Classification of hatchling orientation (adapted from CANBOLAT and NALBANTOGLU, 2001). Hatchlings showing an initial heading direction towards the sea are counted as "oriented" (black arrows). Hatchlings heading in the opposite direction landwards are accounted as "disoriented" (red arrows). Hatchlings heading first landwards, but then changing direction and heading to the sea later, are assessed according to their initial heading direction in the circle and still counted as disoriented. The two examples show typical hatchling tracks in the presence of artificial lights (lamp symbols). The darker grey scale value indicates lower light pollution in the field (see Figure 4.16). A: disorientation, B: misorientation. Hatchlings heading straight landwards towards artificial lights are termed "misoriented", which is the extreme case of disorientation (WITHERINGTON and MARTIN, 1996). Note that in the binominal statistics used in my thesis these hatchlings are counted as disoriented.

Hatchling disorientation per kilometre beach length was calculated for Sections I and II (Appendix 2) and compared per beach sections as defined in this thesis (Tables 4.3). This data was reconciled with the Artificial Light Source Density (ALSD) in given area (Figure 4.16). Statistical evaluation was made according to table 4.8.

Table 4.8: Statistical tests to assess hatchling disorientation in Belek, Sections I and II (2005)

Test name	Assessment[1]	H_0	H_1
Student's t-test	Is hatchling disorientation equal in Sections I and II?	hatchling disorientation is <u>equal</u> in the two beach areas	hatchling disorientation is <u>unequal</u> in the two beach areas
SPEARMAN`s rank correlation test	Is there positive correlation between the Artificial Light Source Density (ALSD) and hatchling disorientation per km?	There is <u>no correlation</u> between the ALSD and hatchling disorientation per km	There is a <u>positive correlation</u> between the ALSD and hatchling disorientation per km

[1] Probabilities for the Student's t-test were assessed one-tailed, for the SPEARMAN's test two- tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$.

Hatchling disorientation and moon phases

Hatchling disorientation is known to be dependent on ambient light. In dark nights at new moon, hatchlings are subject to higher disorientation, caused by artificial lights, compared to full moon nights (SALMON and WITHERINGTON, 1995).

The purpose of my study was to find out if dark nights around new moon increased hatchling disorientation in Belek, and if this was consistent within Sections I and II, which had different levels of overall light pollution.

The SPEARMAN's rank correlation test was used to assess if hatchling disorientation events in Sections I and II were significantly correlated with dark ambient light at new moon phases (Table 4.10). A disorientation event is understood as at least one disoriented hatchling per night, assessed on hatchling tracks found near the nest in the morning (see Figure 4.4 and Appendix 2). Disorientation events per night were summed and the total number of disoriented hatchlings calculated versus the number of days till the closest new moon phase (see example below). The percentage of disoriented hatchlings per night was also calculated. The outcomes of the SPEARMAN's rank correlation test are illustrated in Figures 4.19 a, b.

Table 4.9: Moon phases during the nesting season in Belek, 2005[1)]

New moon	First quarter	Full moon	Last quarter
June 6 (21.55)	June 15 (01.22)	June 22 (4.14)	June 28 (18.23)
July 6 (12.02)	July 14 (15.20)	July 21 (11.00)	July 28 (03.19)
August 5 (03.05)	August 13 (02.38)	August 19 (17.53)	August 26 (15.18)
September 3 (18.45)	September 11 (11.37)	September 18 (02.01)	September 25 (06.41)

[1)] Moon phases in June, July, August and September 2005. These are periods in which *C. caretta* hatchling emergences were recorded in Belek. The exact time (UTM) of given moon phase incidence is indicated in brackets[22].

Table 4.10: Assessment of hatchling disorientation at different moon phases (SPEARMAN's rank correlation test)

Test name	Assessment[1)]	H$_0$	H$_1$
SPEARMAN's rank correlation test	Is there a positive correlation between new moon phases and increased hatchling disorientation?	There is no correlation between new moon phases and increased hatchling disorientation	There is a positive correlation between new moon phases and increased hatchling disorientation

[1)] All probabilities were assessed two-tailed. Significance level for H$_1$: $p < 0.05$, high significance level: $p < 0.005$. Statistics were conducted using VassarStats statistical computation software.

Example

In Section I, a total 28 hatchlings were disoriented in the night of August 8th, which is 3 days from the closest new moon on 5th August 2005 (Table 4.9). In the same area there were no disoriented hatchlings found in the night of August 29th, which is 6 days from the closest new moon on 3rd September 2005. Resulting observation pairs for use in SPEARMAN's rank correlation test:

x (number of days to closest new moon)	y (disoriented hatchlings at given date)
3 (days from the closest new moon on 5th August)	**28**
6 (days from the closest new moon on 3rd September)	**0**

Section I (monitoring area AY): In total 46 emergence events were evaluated in 30 nights, with a total of 311 disoriented hatchlings counted in 34 nests.
Section Il (monitoring areas AOY and OY): In total 70 emergence events were evaluated in 30 nights, with a total of 1096 disoriented hatchlings counted in 43 nests.

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[22] http://aa.usno.navy.mil/data/docs/MoonPhase.php

Hatchling mortality

Based on the number of dead hatchlings found on the beach (see Appendix 2), statistical evaluation was made for hatchling mortality, according to Table 4.11.

Table 4.11: Statistical tests to assess hatchling mortality in Belek, Sections I and II (2005)

Test name	Assessment[1]	H_0	H_1
Student's t-test	Is hatchling mortality equal in Sections I and II?	hatchling mortality is <u>equal</u> in the two beach areas	hatchling mortality is <u>unequal</u> in the two beach areas
SPEARMAN`s rank correlation test	Is there a positive correlation between hatchling disorientation and hatchling mortality?	There is <u>no correlation</u> between hatchling disorientation and hatchling mortality	There is a <u>positive correlation</u> between hatchling disorientation and hatchling mortality

[1] Probabilities for the Student's t-test were assessed one-tailed, for the SPEARMAN's test two-tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$.

The outcomes of the statistical tests on nest density, hatchling disorientation and hatchling mortality are given in Chapter 4.2.

4.1.5 Hatchling orientation experiments in the field

I conducted experiments with a small sample size of 10-19 *C. caretta* hatchlings, which were available from two nests, to test orientation behaviour in the presence of strong artificial lights and anthropogenic noise (Figure 4.20). Whereas the negative impact of artificial lights has been proven in experiment, little is known about the effect of low frequency sounds on hatchling behaviour. The objective of this study was to find out how hatchlings reacted in an experimental setup to the presence of simultaneous acoustic and visual stimuli, under real conditions in the egg-laying zone of Belek.

Experimental setup

C. caretta hatchlings were taken out of two nests in the late afternoon, shortly before emergence, and kept in the dark at a cool place until night. After nightfall, which is the natural emergence time of hatchlings (WITHERINGTON et al., 1990), they were put in the experimental arena (80 x 80 x 5 cm), containing a thin layer of sand (1-2 cm), to imitate natural crawling conditions. The low profile of this construction enabled visual orientation in hatchlings. The wooden construction of the area was chosen to transmit substrate vibrations caused by the surf. The experimental field was set up in the Public area 3 of the Tourism Development Area (see Table 4.3 b, Appendix 3 f), 26 m from the surf zone in the south, and 50 m from a discotheque, which emitted strong airborne noise from the east. Light pollution was being emitted from the same discotheque (Metal Halide lights) and also from a coastal street (High Pressure Sodium Vapour lights) located 80 m northwest of the experimental arena.
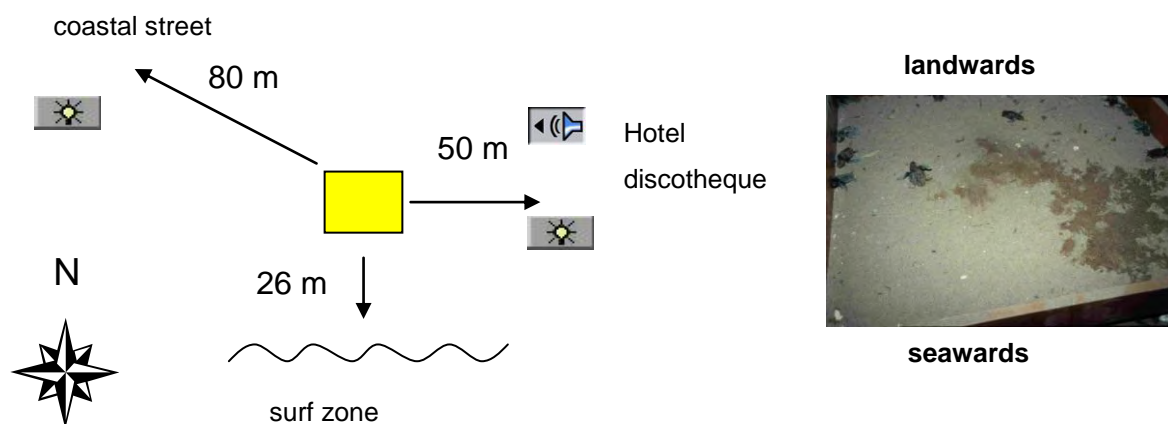
Figure 4.5 a: Experimental setup used for testing *C. caretta* in the presence of artificial lights (lamp symbol) and noise (speaker symbol).

Figure 4.5 b: Experimental arena with hatchlings orienting to its periphery.

Of four experimental setups used, lights were excluded in Setup 1 and Setup 2 by covering the area with a wooden board. Substrate vibrations were excluded in Setup 2 and 4 by air-shielding the bottom of the box. Strong airborne sounds from the hotel discotheque were present constantly. Groups of 10-19 hatchlings per setup were put in the centre of the box facing inwards. Crawling behaviour was observed (Setups 3 and 4). Observation was not feasible in Setups 1 and 2 due to the light cover used. The final position of the hatchlings within the box was recorded after 10 minutes (all setups).

Table 4.12: Experimental setup using *C. caretta* hatchlings in Belek

Experimental Setup	Lights	Substrate vibrations	Airborne sounds
Setup 1 (n=10)	-	+	+
Setup 2 (n=19)	-	-	+
Setup 3 (n=17)	+	+	+
Setup 4 (n=16)	+	-	+

Plus and minus symbols indicate the presence of stimuli which were adapted for different setups. Note that airborne sounds were present constantly.

Statistics

The Sign test was used to evaluate the distribution of hatchlings in one setup. It was tested if hatchlings significantly oriented in a preferred direction, seawards or landwards (H_1 = true), or if they were equally distributed (H_0 = true). The FISHER'S Exact Probability test was used to compare the distribution of hatchlings in two setups. It was tested if hatchlings showed significant directional orientation. This was distribution in the lower half (seawards) or upper half (landwards) of the experimental arena (Figure 4.5 b).

Table 4.13: Statistics used to evaluate *C. caretta* hatchling behaviour in Belek

Test name	Assessment[1]	H$_0$	H$_1$
Sign test	Assessing binominal distribution of hatchlings in the experimental arena for <u>one</u> experimental Setup	Hatchlings are <u>equally</u> distributed in the experimental arena	Hatchlings are <u>unequally</u> distributed in the experimental arena
FISHER'S Exact Probability test	Comparing distribution of hatchlings in the experimental arena for <u>two</u> experimental Setups	Hatchlings <u>equally</u> distributed in two experimental Setups.	Hatchlings are <u>unequally</u> distributed in two experimental Setups

[1] All probabilities were assessed two-tailed. Significance level for H$_1$: $p < 0.05$, high significance level: $p < 0.005$. Statistics were made using VassarStats statistical computation software.

The outcomes of the experiments are given in Chapter 4.2.

4.2 Results

4.2.1 Progression of coastal development in Belek, from 2000-2004

ASTER satellite data revealed that there was a considerable progression of coastal development in the Tourism Development Area (Section II) from July 2000 to July 2004, whereas the Natural Site Area (Section I) remained undeveloped to a large extent (Figures 4.6 a, b). In Section II there was a hotel complex with six new hotels built during this period. From west to east this is "Sillyum", finished end of 2000, "Cornelia" in 2002, "Pine Beach" in 2001, "Atlantis" in 2001/2002, "Gloria Verde" in 2002 and "Zeugma" in 2002. For the entire Section II this is a considerable increase compared with 23 hotels in 2000 to 29 hotels in 2004. This equals 0.98 km more beach length covered with buildings or an increase of 13.8 percent in coastal development (hotels) in Section II from 2000 to 2004. The satellite data are consistent with my observations in the field. Section I showed a relatively low degree of coastal development in September 2005. Hotels or major condominiums were not found on its 4.8 km beach length. Some coastal development was evident east of the Aksu River. This was a small settlement, which was located along the Aksu River, and about 30 small huts located between the Aksu and Besgöz estuaries close to the surf zone. Fishermen inhabited these during the summertime. In contrast, there was a high degree of coastal development west of the Aksu River (Figure 4.7), which was outside my study area. In Section II there was also high-degree coastal development on its entire 11.3 km beach length, with 29 hotels and associated infrastructure located within 100 m of the shore (Figures 4.8, 4.9). This also included water sport areas and discotheques, two construction sites (Figure 4.10), which were also operative at night, and a coastal street that was located 63 m from the shoreline (Figure 4.15).

The results confirm that this important *C. caretta* nesting site in Belek was under intense coastal development in recent years.
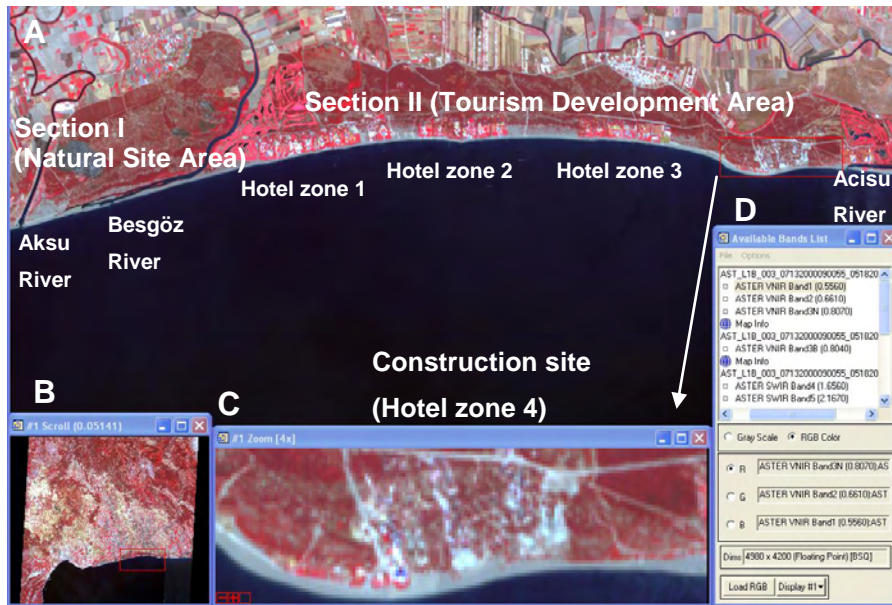
Figure 4.6 a: Coastal development in Belek, **July 2000** (A). The overview window (B) shows the location of Belek on the Gulf of Antalya. The zoom window (C) shows the construction site, which is the later Hotel zone 4. The construction site is clearly distinguishable by its greyish colour from surrounding vegetated areas. These appear red in the RGB false colour projection setting (D).
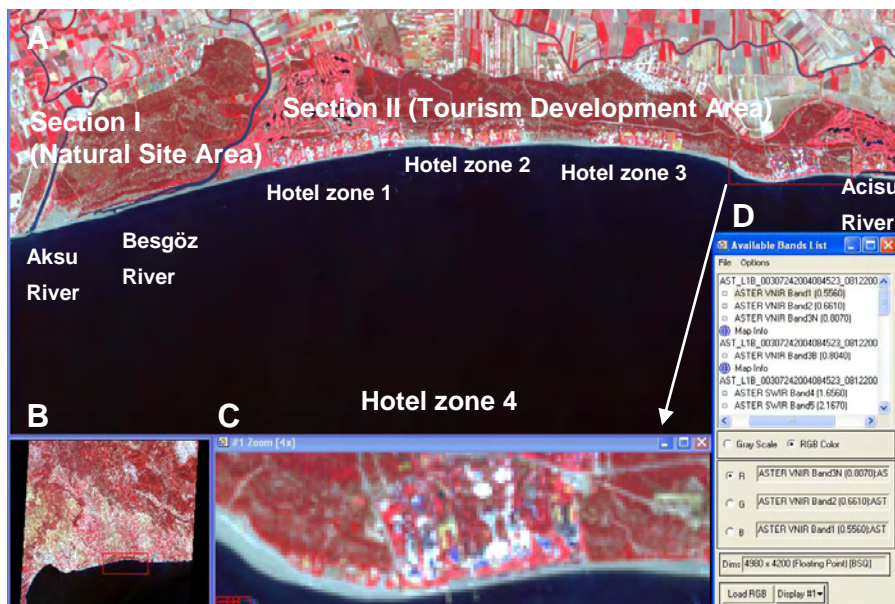
Figure 4.6 b: Coastal development in Belek, **July 2004** (A). The zoom window (C) shows the Hotel zone 4, which was finished in Section II from 2000 to 2002. Buildings (hotels) appear white, whereas water (swimming pools) appears blue and vegetation (green spaces, parks, golf areas) appears red. Note also an increase in golf areas north of the Acisu River and Hotel zone 1, replacing pine forest (dark red)

Figure 4.7: Coastal development at Aksu River mouth (Section I)

Figure 4.8: Hotel beach (Section II)

Sand compaction by human activity

Figure 4.9: Water sport area (Section II).

Sand compaction by vehicles

Figure 4.10 a: Construction site (Section II).

Sand-mining. In the front: sea turtle nest

Figure 4.10 b: Construction site (Section II).

Waste on the beach, in front of sea turtle nest

4.2.2 Coastal development and light pollution in Belek (2005)

The terms sky glow, direct irradiation and Artificial Light Source Density (ALSD) used here are defined in Chapter 4.1. The subdivisions used for Sections I and II are according to Tables 4.3 a, b.

Observation in the field revealed that there was light pollution detectable in the entire study area (Sections I and II, Figure 4.16). In <u>Section I</u> light pollution was caused by sky glow. This was propagated from the nearby settlements, Buyukkumluca in the northwest (lat: 36.878, long: 30.907) and Belek in the northeast (lat: 36.863, long: 31.056), and from the adjacent hotel zones. Section I was also considerably affected by direct irradiation from artificial lights, which were set up in the hotel zones and were even clearly detectable from dark areas (Figure 4.11). Light pollution was particularly high in the western area of Section I. This was caused by a minimum of 14 Metal Halide lights, which were west of Section I, close to the Aksu River, and shining eastwards (see Appendix 3 a). This highlights a considerable problem for sea turtle hatchlings emerging in this area of Section I, which was confirmed by increased disorientation even in > 2 km distance to the light sources (see chapter 4.3.4). Fishing boats caused comparatively low levels of light pollution along the Besgöz River, which was the central area of Section I. The eastern area of Section I was also affected by direct irradiation. This was caused by high intensity Metal Halide lights, which were at the Hotel Asteria (Section II) and shining westwards (Appendix 3 b).

<u>Section II</u> showed high levels of light pollution on its entire 11.3 km length, caused by sky glow and direct irradiation from the hotel zones 1-4. Light pollution was congruent with intensive coastal development in this area. Major light sources were a minimum of 57 Metal Halide lights that were positioned close to the hotels and construction sites and were clearly visible from the beach (Figures 4.12, 4.13, 4.14). The MH lights were usually directed straight at the beach, for security reasons and to enable night work in the hotel zones. These lights were also visible from the back within a 360° angle, as they had no shielding against stray light. This made the whole beach brightly illuminated at night until at least 4 am, when night work at the hotels and construction sites was finished. There was also a high number of smaller light sources (by wattage) set up in the hotel zones, including incandescent lights, fluorescent lights, neon tubes and LED lights, which could not be counted due to their multiplicity. Moreover there were 16 High Pressure Sodium Vapour (HPS) lights positioned along the coastal street (Figure 4.15), which were operated all night long.

Though HPS lights were directed downwards for illuminating the street, there was considerable irradiation of the nesting beach due to the poor shielding properties of the lamp's "shoebox" fixture. The problem of light trespass was amplified due to the high mounting of the bulbs (10 m) and relative closeness to the shoreline (63 m). Both MH and HPS lights have polychromatic characteristics, emitting in a broad spectral range (see Chapter 1) and known to be highly disruptive for females and hatchlings (WITHERINGTON and MARTIN, 1996). Therefore they must be considered unacceptable for the *C. caretta* index nesting beach in Belek.

Artificial Light Source Density (ALSD)

There were no major light sources set up in Section I. Consequently the ALSD for the entire Section I (4.8 km) is zero. In contrast, the ALSD was high in Section II, mainly in the hotel zones (see Appendix 3 c-f). In the hotel zones 1 and 2, stationary Metal Halide (MH) lights accounted for an ALSD of 7.8 and 6.3 respectively. In the hotel zones 3 and 4, major stationary MH lights accounted for an ALSD of 5.1 and 9 respectively. In contrast, the ALSD was low in the public areas, with an ALSD of 2 MH lights in Public area 1, and 1.5 MH lights in Public area 2. There were no MH lights set up in Public area 3, but High Pressure Sodium Vapour (HPS) lights (ALSD: 8.7) along the coastal street. The ALSD for smaller stationary light sources set up in Section II, including incandescent lights, fluorescent lights, neon tubes and LED lights, could not be calculated due to their multiplicity. The results on ALSD for MH lights and HPS lights are summarized in Figure 4.16.

Illuminance measurements

Measurements in the field revealed highly variable illuminance in the egg-laying zone, which was generally lower in Section I compared with Section II. Highest illuminance (E_{max}) was measured in Section II in hotel zones, with a high ALSD of high intensity Metal Halide lights up to 1000 watts (see Figure 4.14, Appendix 3 f). Lowest illuminance (E_{min}) was measured in the central area of Section I, which was the darkest part of the study area and lacking close artificial light sources (ALSD = 0) (see Appendix 3 a, b).
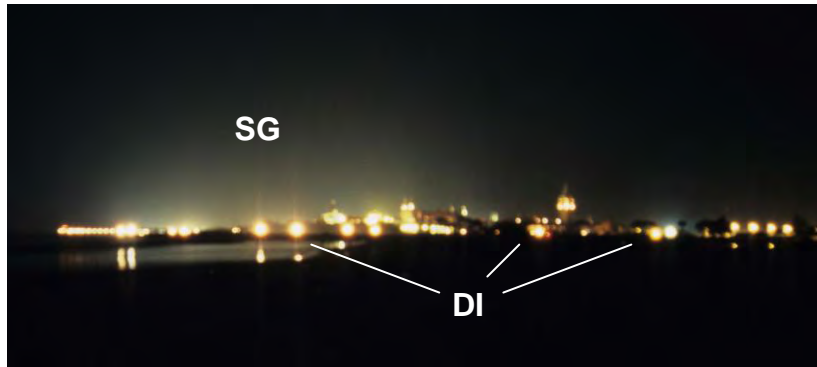
Figure 4.11: Direct irradiation (DI) and sky glow (SG) at Aksu River mouth (Section I)
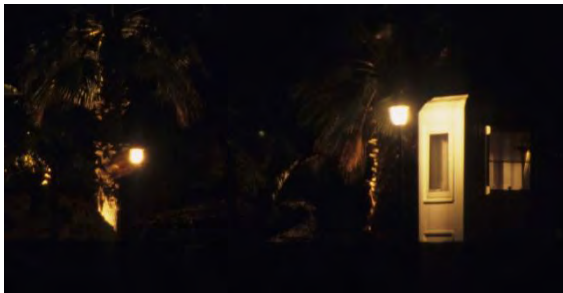
Figure 4.12: Metal Halide (MH) floodlights in the hotel zone (Section II). Mounting height: 2 m
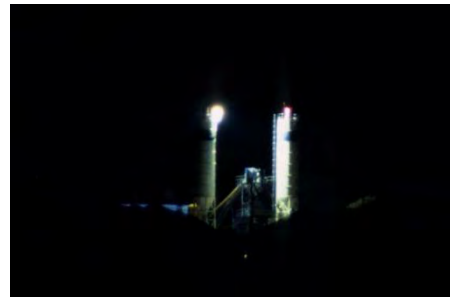
Figure 4.13: MH floodlight at construction site (Section II). Mounting height: about 15 m

Figure 4.14: MH light in close proximity to a protected sea turtle nest (Section II). Mounting height: 8 m

Figure 4.15: High Pressure Sodium Vapour (HPS) and car lights on coastal street (Section II). Mounting height of HPS lights: 10 m

Table 4.14: Illuminance measured in the study area of Belek

Source (wattage, colour)	Distance from source	Illuminance (E_V) [1]	location
Nocturnal light (near full moon)	384 401 km (moon-earth average)	0.11 lux	Section I, Central area
Nocturnal light (half moon)	384 401 km	0.07 lux	Section I, Central area
Nocturnal light (new moon)	384 401 km	0.05 lux (E_{min})	Section I, Central area
HPS light (Lucalox 250 watts, yellow-white)	30 m	1.28 lux	Section II, Public area 3
Metal Halide light (70 watts, white)	20 m	6.0 lux	Section II, Hotel zone 2
Metal Halide light (1000 watts, white) (Beta and Delta Projectors)	10 m	39.8 lux (E_{max})	Section II, Hotel zone 4

[1] All measurements were made in the egg-laying zone close to sea turtle nests. Note that values represent overall illuminance, which is a result of ambient light and the luminous intensity of artificial lights in areas affected by light pollution. E_{max}: highest, E_{min}: lowest illuminance measured.

	West	Centre	East	Hotel zone 1		Hotel zone 2		Hotel zone 3		Hotel zone 4
					Public area 1		Public area 2		Public area 3	
Coastal development	(-)	-	-	+	(+)	+	(+)	+	(+)	+
ALSD (model)	0	0	0	7.8 (MH)	2 (MH)	6.3 (MH)	1.5 (MH)	5.1 (MH)	8.7 (HPS)	9 (MH)
Illuminance (lux)	< 0.11			< 40	< 6	< 40	< 6	< 40	< 6	< 40
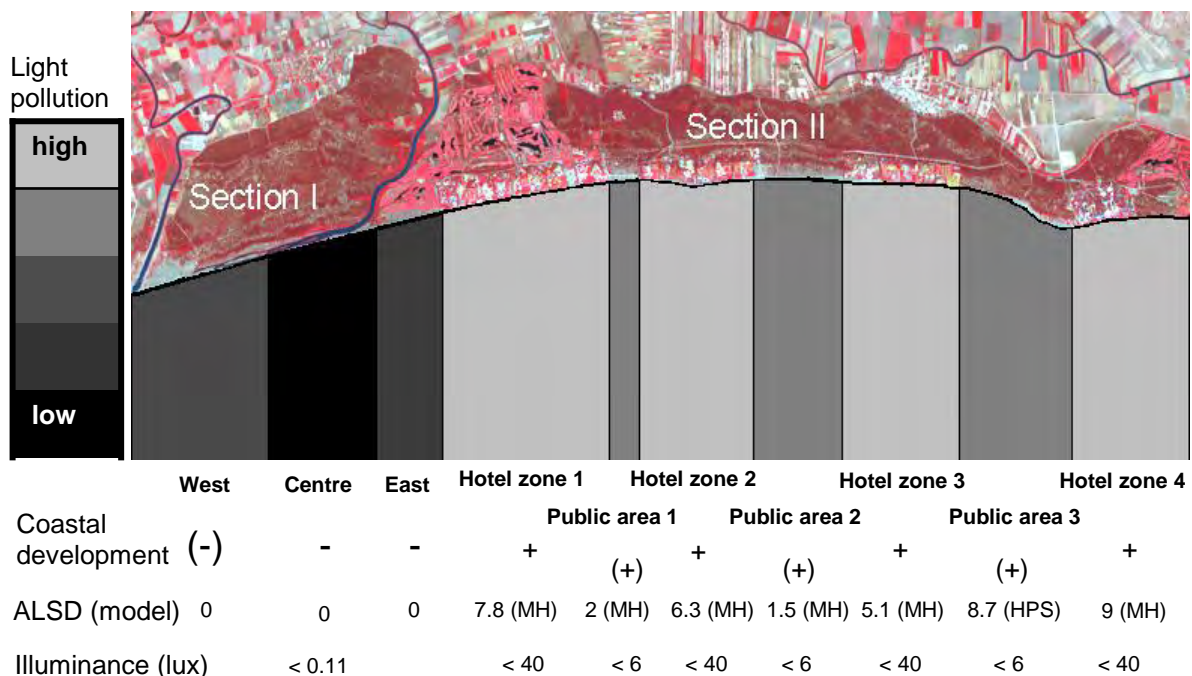
Figure 4.16: Coastal development and light pollution in Belek, Sections I and II (2005). Symbols indicate the degree of coastal development: + = highest, - = lowest. Artificial Light Source Density (ALSD): Metal Halide (MH), High Pressure Sodium Vapour (HPS), illuminance as defined in Table 4.14

4.2.3 Ambient sounds and noise pollution

Measuring surf sounds (1 m wave height) with the HMB-TEC A316 revealed sound levels of up to 82 dB (SPL) in proximate distance to the source. There was a continuous attenuation in SPL from 5 m to 25 m distance from the shoreline, with 72 dB (SPL) measured at 25 m distance (Table 4.15, 1a-e). The frequency curves of the surf sounds measured remained constant with increasing distance, with distinctive peaks in the low frequency range < 500 Hz (Figure 4.18).

In Belek, noise pollution was evident. This was mainly caused by stationary discotheques, located in front of the hotels, but also by mobile dance floors that were built up close to the shore in the public areas at night. Some discotheque noise was measured to be 81 dB (SPL) at 1 m distance (Table 4.15, 3b). In the presence of this source, 47 dB (SPL) were measured at 50 m distance, where several sea turtle nests were found (Table 4.15, 3.a). Another discotheque emitted up to 95 dB (SPL) at 1 m distance (Table 4.15, 2b). In the presence of this source, 55 dB (SPL) were measured at 100 m distance, where nests were found (Table 4.15, 2a). Frequency analysis revealed that discotheque sounds had peaks in the very low frequency spectrum < 100 Hz. This corresponds with the high intensity low-bass component of modern dance music, such as "Techno" and "R'n'B". These low frequency sounds were propagated over several hundred meters and could be physically perceived as vibrations on the beach. Frequency peaks were also found in the mid-range spectrum at about 1250 Hz and higher, due to the harmonics of the high component of the dance music. Other major noise sources measured in the field were fireworks, which could not be measured directly at the source. However, these fireworks clearly increased overall noise levels in the egg-laying zone, with up to 74 dB (SPL) measured at an estimated 50 m distance (Table 4.15, 3c). Using Equation 4.3 for point sources, fireworks were measured as emitting up to 107 dB SPL at 1 m from the source. This is probably one of the highest sound levels of anthropogenic noise in this nesting area. The frequency spectra of fireworks could not be measured due to the short periodicity of this noise source. Another major source identified in the egg-laying zone was car engine noise (Table 4.15, 4b), with sound levels of up to 55 dB (SPL) measured 34 m from its source (Table 4.15, 4a). This was the location of sea turtle nests near a coastal street (see Appendix 3 f). Street traffic noise in this area was present 24 hours round the clock, with no substantial diminution observed during the night hours (Figure 4.15). Frequency analysis of car noise revealed peaks at 50 –

3000 Hz, which includes the spectrum perceived by sea turtles (RIDGWAY et al., 1969). Other sources of noise were <u>water sport areas</u> (speed-boats and jet-skis) and <u>construction sites</u> (vehicles and dredging), which were maintained mainly during the day. These noise sources were not investigated in detail, as contributing less to noise pollution in the night, when sea turtles and hatchlings emerged. However, some construction activity like dredging (Figure 4.10 a) extended even into the night.

In overall my measurements indicate that:

1. There is a similarity between the spectra of some noise sources and the low frequency component of natural wave/surf sound at < 1000 Hz.

2. Air-propagated noise in the egg-laying zone may have comparable or higher sound levels as wave/surf sounds.

Table 4.15: Sound measurements with the HMB-TEC A316 in Section II Belek

Measurement[1]	Sound source(s)	Measuring distance	SPL (max.)	Frequency peaks
1a	Surf (about 1 m wave height)	5 m from the surf	82.4 dB(SPL)	< 500 Hz
1b	Surf (1 m)	10 m from the surf	80.4 dB(SPL)	< 500 Hz
1c	Surf (1 m)	15 m from the surf	77 dB(SPL)	< 500 Hz
1d (figure 4.18)	Surf (1 m)	20 m from the surf	74 dB(SPL)	< 500 Hz
1e	Surf (1 m)	25 m from the surf	71.8 dB(SPL)	< 500 Hz
2a	Surf (about 0.5 m wave height) plus discotheque A	22 m from the surf 100 m from noise source	55.3 dB(SPL)	< 50 and > 1250 Hz
2b	Discotheque A	1 m from noise	95.3 dB(SPL)	< 100 and > 1250 Hz
3a	Surf (0.5 m) plus discotheque B	26 m from the surf 50 m from noise	47.3 dB(SPL)	< 50 and > 1250 Hz
3b	Discotheque B	1 m from noise	81 dB(SPL)	< 50 and > 1250 Hz
3c	Surf (0.5 m) plus discotheque B plus fireworks	26 m from the surf 50 m from noise	73.5 dB(SPL)	< 50 and > 1250 Hz
4a	Surf (0.5 m) plus car noise	26 m from the surf 34 m from noise	55 dB(SPL)	100 – 3000 Hz
4b	car noise	1 m from noise	85.1 dB(SPL)	50 – 3000 Hz

[1] Measurements 1a-e show the sound level attenuation of surf sound with increasing distance from the source. Note that the sound level of the surf is dependent on its wave height. Measurements on noise 2b, 3b and 4b were made directly at the source; 2a, 3a, 3c and 4a were made in proximate distance to sea turtle nests. Consequently the SPL measured is a summation of various broadband sources in the field here.
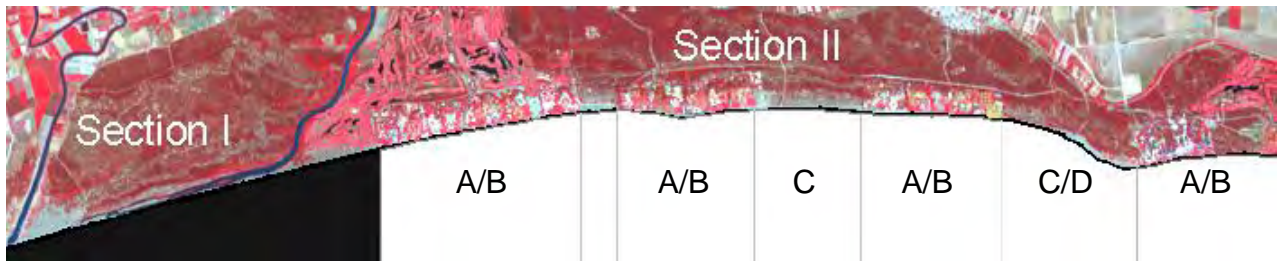
Figure 4.17: Noise pollution in Section II, Belek (2005). A: Water sport areas (speed-boats, jet-skis), B: Hotels (fireworks, discotheques), C: Construction sites (vehicles, dredging), D: Coastal roads (cars, tractors, beach vehicles). Noise pollution was nearly absent in Section I (Natural site area).

Auditory data correlation

Four hundred Hz is the range in which juvenile *C. mydas* perceive aerial surf sounds best, with 1 KHz being the upper perception limit (RIDGWAY et al., 1969; see Figure 1.2). Based on these data, I calculated that low frequency surf sounds (1 m wave height), recorded on the beach at 20 m distance from the source, are generally in the hearing range of sea turtles (see Tables 4.5, 4.6). There is a remarkable consistency in the run of the frequency curve of surf sound and juvenile *C. mydas* auditory sensitivity between 50 Hz and 400 Hz. In the higher spectrum both curves diverge and overlap at 700 Hz (Figure 4.18). This means, at the given surf intensity and distance, that the surf spectrum > 700 Hz is outside the hearing range of *C. mydas*.
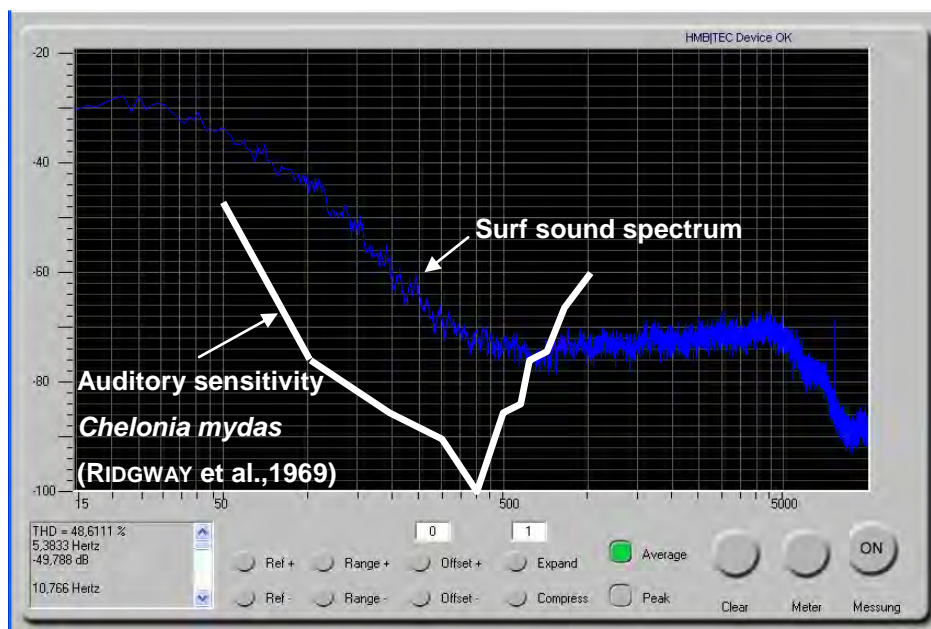
Figure 4.18: Comparison of the surf sound spectrum, measured at a sea turtle nest 20 m from the shoreline, and the auditory sensitivity of juvenile *C. mydas*, measured by Ridgway et al. (1969), transformed to V-scale (see Table 4.6). The graphs show that the low frequency component of surf sounds from 50 Hz to 700 Hz is in the hearing range of this species. Abscissa: frequency (Hz), ordinate: output voltage of the Audio Frequency Analyser (dBV).

4.2.4 Female nesting density, hatchling disorientation and mortality

Nesting density

In the 2005 nesting season, a total of 145 nests were counted in Section I during the beach patrols on 4.8 km beach length. In Section II, a total of 101 nests were found on 11.3 km beach length (CANBOLAT, pers. com.). GPS data evaluation revealed that Section I (Natural Site Area) had a higher nesting density, with 30 *Caretta caretta* nests per km, compared with Section II with only 9 nests per km.

In Section I, there was higher nesting density in its western and central area along the Besgöz River, whereas fewer nests were found in its eastern area, which was adjacent to the hotel "Asteria" in Section II (see Appendix 3 a, b).

In Section II, there was considerably low nesting density in front of hotels (4 nests/km) compared to public areas (18.5 nests/km), which were not yet covered with buildings or under construction in 2005 (see Appendix 3 c-f).

Table 4.16: Nesting densities in Sections I and II, Belek (2005 nesting season)

Location	Nesting density[1]
Section I	**mean: 30**
west	29.5
centre	33.5
east	24.5
Section II	**mean: 9**
Hotel zone 1	3.5
Public area 1	4
Hotel zone 2	3
Public area 2	20
Hotel zone 3	8.5
Public area 3	19.5
Hotel zone 4	2.5

[1] Calculation is based on the nest number per 1 km beach length, according to Tables 4.3 a, b. The nest coordinates were obtained by GPS (CANBOLAT, with permission)

The Student's t-test revealed significance of H_1 (nesting densities are unequal in two beach areas) for all tests conducted (Table 4.17). The outcomes show that there is a significant difference in nesting densities, with Section I (Natural Site Area) having higher nesting densities compared with Section II, and public areas in Section II having higher nesting densities compared with hotel zones in Section II.

Table 4.17: Statistical outcomes for nesting densities (ND) in Belek (2005 nesting season)

Student's t-test (Comparing means of nesting densities)	Section I vs. Section II (public areas)	Section I vs. Section II (hotel areas)	Section II (public areas) vs. Section II (hotel areas)
Result[1]	ND (Section I) > ND (Section II, public areas) (H$_1$ with p = 0.033)	ND (Section I) > ND (Section II, hotel areas) (H$_1$ with p = 0.00014)	ND (Section II, public areas) > ND (Section II, hotel areas) (H$_1$ with p = 0.042)

[1] All probabilities were assessed one-tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$. The calculations are based on the means of nesting densities provided in table 4.16.

The SPEARMAN's rank correlation test revealed a highly significant negative correlation between the Artificial Light Source Density (ALSD, see Chapter 4.2.2, Figure 1.16) and the nesting density [H_1 with p (two-tailed) = 0.001]. These outcomes show that the nesting density is significantly reduced in areas of high ALSD.

Hatchling disorientation

In Section I (Natural Site Area), of a total of 145 nests that were found here in 2005, 34 nests showed hatchling disorientation (n= 311). This must be considered a minimum value, as data was deficient for 36 nests. In Section II, of a total of 101 nests, 43 nests showed hatchling disorientation (n= 1096). This is also a minimum value, as data was deficient for 23 nests. These results indicate that disorientation is considerably more frequent in Section II than in Section I. In **Section I**, disorientation was evident in its western area, with 119 disoriented hatchlings found in 16 nests near the Aksu and Besgöz estuaries (Appendix 3 a). This is 47% of the total hatchling disorientation in Section I. In the central area, there were 186 disoriented hatchlings found in 17 nests along the Besgöz River. This is 50% of the total hatchling disorientation in Section I. In the eastern area, close to the hotel "Asteria", there were only 6 disoriented hatchlings found in one nest (Appendix 3 b). This is

remarkable, as light pollution was higher here compared with the central area of Section I (Figure 4.16). In **Section II**, of 1096 disoriented hatchlings found in 43 nests, 609 hatchlings from 19 nests were disoriented in front of hotels (Hotel zones 1-4, Appendix 3 c-f). This is 55.6% of the total hatchling disorientation in Section II. 487 disoriented hatchlings from 24 nests were found in the Public areas 1, 2 and 3, which is 44.4%. Taking the beach length (Table 4.3 b) into account, my statistics reveal that hatchling disorientation per km was generally higher in public areas (average 131.6) than in hotel zones (average 80.1). This is also remarkable, as light pollution was generally assessed as higher in the hotel zones compared with public areas.

Table 4.18: Hatchling disorientation in Sections I and II of Belek, (2005 nesting season) [1]

Location	Hatchlings emerged[2]	Hatchlings disoriented[1]	Hatchlings disoriented (%)	Hatchlings disoriented per km[3]
Section I (Natural Site Area)	5655 (in 145 nests)	311 (in 34 nests)	5.5 (average for entire Section I)	54.7 (average for entire Section I)
West	1911	119	6.23	71.6
Centre	2691	186	6.91	86.5
East	1053	6	0.57	6.1
Section II (Tourism Development Area)	3822 (in 78 nests)	1096 (43 nests)	28.7 (average for entire Section II)	97 (average for entire Section II)
Hotel zone 1	454	310	68.3	120.6
Public area 1	55	0	0	0
Hotel zone 2	290	109	37.6	62.6
Public area 2	1060	264	25	197
Hotel zone 3	691	86	12.4	48.9
Public area 3	1130	223	19.7	121.2
Hotel zone 4	142	104	73.2	66.7

[1] Based on monitoring data obtained by CANBOLAT (with permission, see Appendix 2).
[2] Note that data for Section I are estimates due to high data deficiency (see Appendix 3 a, b). In Section II data are not available for 23 nests.
[3] Calculations are based on monitoring data and comparison with beach lengths (Table 4.3 a, b)

The Student's t-test revealed no significance of H_1 (hatchling disorientation is unequal in two beach areas) for the tests conducted (Table 4.19). Therefore it cannot be shown that hatchling disorientation is significantly increased in Section II (hotel zones), compared with Section II (public areas) and Section I (Natural Site Area).

Table 4.19: Statistical outcomes for hatchling disorientation (HD) in Belek (2005 nesting season)

Student's t-test (Comparing hatchling disorientation per km)	Section I vs. Section II (public areas)	Section I vs. Section II (hotel areas)	Section II (public areas) vs. Section II (hotel areas)
Results[1]	HD (Section II, public areas) = HD (Section I) (H_1 with p = 0.229133)	HD (Section II, hotel areas) = HD (Section I) (H_1 with p = 0.251883)	HD (Section II, hotel areas) = HD (Section II, public areas) (H_1 with p = 0.284259)

[1] All probabilities were assessed one-tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$. The calculations are based on hatchling disorientation per km provided in Table 4.18.

The SPEARMAN's rank correlation test did not reveal a significant correlation between the Artificial Light Source Density (ALSD, Figure 4.16) and the total number of disoriented hatchlings per beach area [H_1: p (two tailed) = 0.644].
These outcomes show that hatchling disorientation is not increased in areas of higher ALSD. This will be discussed in Chapter 4.3.

Hatchling disorientation and moon phases

In the 2005 nesting season, most hatchling emergences were observed from the end of July to the beginning of September. Hatchling emergence generally happened at regular intervals, independent of the moon phase. Using the monitoring data on hatchling disorientation (Appendix 2), the SPEARMAN's rank correlation test revealed that in Section II (Tourism Development Area) hatchling disorientation was strongly positively correlated with dark ambient light around new moon phases (Table 4.9) [H_1 with p (two-tailed) = 0.000483]. In contrast, in Section I (Natural Site Area) there was no correlation between hatchling disorientation and new moon phases [H_1 with p (two-tailed) = 0.854]. Hatchling disorientation was more constant over the months at different moon phases here.

The outcomes of the analysis are illustrated in Figures 4.19 a, b. Note that hatchling disorientation occurred in both Sections, but the total number of disoriented hatchlings was generally higher in Section II, with up to 150 disoriented hatchlings counted per night here, which was a disorientation rate of 83% (Fig. 4.19 b).
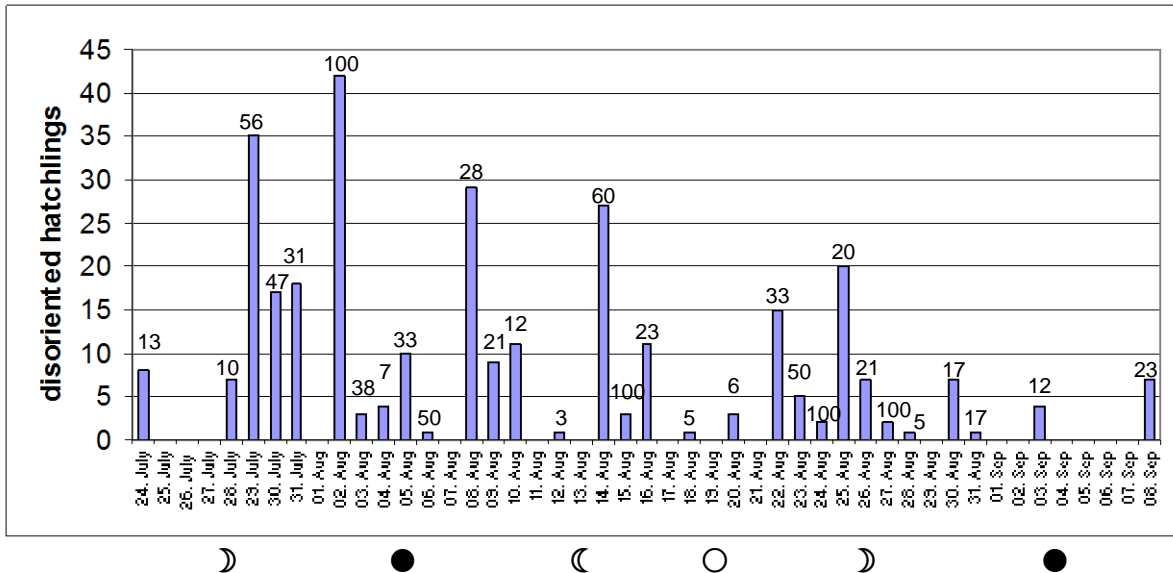
Figure 4.19 a: Total number of disoriented hatchlings per moon phase in Belek, **Section I** (2005). Numbers at the top of the bars indicate the percentage of disoriented hatchlings per night. Symbols on the abscissa represent the moon phases at given dates: new ●, first quarter ☾, full ○, last quarter ☽. Note that hatchling disorientation is nearly uniform over the lunar cycle.
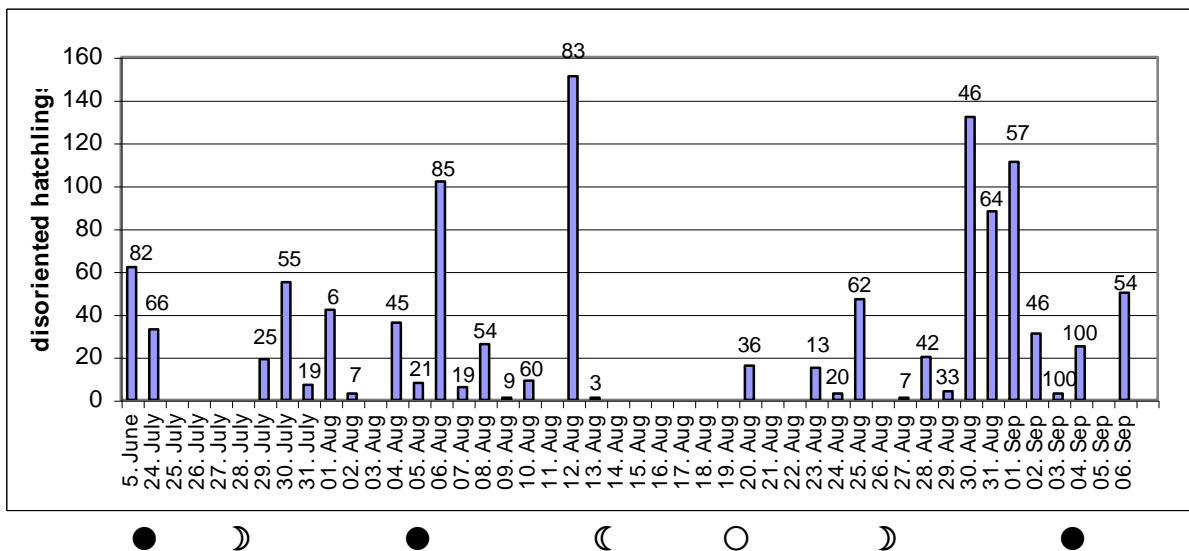
Figure 4.19 b: Total number of disoriented hatchlings per moon phase in **Section II** (2005). Numbers at the top of the bars indicate the percentage of disoriented hatchlings per night. Symbols on the abscissa represent the moon phases at given dates: new ●, first quarter ☾, full ○, last quarter ☽. Note increased hatchling disorientation around new moon phases.

Hatchling mortality

In **Section I**, of a total of 1351 hatchlings from 35 nests which were evaluated, 1040 reached the sea and 234 were found dead on the beach. Of these dead hatchlings, 152 (65%) were "disoriented" per definition (see Figure 4.4) and 82 hatchlings (35%) showed initial seaward orientation and so counted as "oriented". In **Section II**, of a total of 3822 hatchlings from 78 nests which were evaluated, 3685 reached the sea and 137 were found dead on the beach. Of these dead hatchlings, 109 (79.6%) were disoriented and 28 (20.4%) showed initial orientation seawards.

Table 4.20: Hatchling mortality in Sections I and II, Belek (2005 nesting season)

Location	Hatchlings emerged	Hatchlings found dead (mortality)[1]	"Disoriented" hatchlings found dead	"Oriented" hatchlings found dead
Section I	1351 (in 35 nests)	234 (17% of all hatchlings emerged)	152 (65%)	82 (35%)
West	545	131	89	42
Centre	763	103	64	39
East	43	0	0	0
Section II	3822 (in 78 nests)	137 (3.6% of all hatchlings emerged)	109 (79.6%)	28 (20.4%)
Hotel zone 1	454	0	0	0
Public area 1	55	0	0	0
Hotel zone 2	290	6	6	0
Public area 2	1060	23	6	17
Hotel zone 3	691	11	11	0
Public area 3	1130	43	43	0
Hotel zone 4	142	54	54	0

[1] Data on hatchling mortality are considered minimum values as there is data deficiency for nests, especially in Section I (see Appendix 3 a, b). Moreover, only hatchlings found dead on the beach were assessed. This probably does not reflect the effective mortality due to predation on land.
Note that in Section I (Natural Site Area) hatchling mortality was nearly equal in disoriented and oriented hatchlings. In contrast, hatchling mortality was proportional to the number of disoriented hatchlings only in Section II (Tourism Development Area).

The Student's t-test revealed no significance of H_1 (hatchling mortality is unequally high in two beach areas) for the tests conducted (Table 4.21). Therefore it cannot be shown that hatchling mortality is significantly increased in Section II (hotel zones), compared to Section II (public areas) and Section I (Natural Site Area).

Table 4.21: Statistical outcomes for hatchling mortality (HM) in Belek (2005 nesting season)

Student's t-test Comparing means of hatchling mortality in	**Section I** with **Section II** (public areas)	**Section I** with **Section II** (hotel areas)	**Section II** (public areas) with **Section II** (hotel areas)
Results[1]	HM (Section II, public areas) = HM (Section I) (H$_1$ with p = 0.1482)	HM (Section II, hotel areas) = HM (Section I) (H$_1$ with p = 0.1021)	HM (Section II, hotel areas) = HM (Section II, public areas) (H$_1$ with p = 0.4173)

[1] All probabilities were assessed one-tailed. Significance level for H_1: $p < 0.05$, high significance level: $p < 0.005$. The calculations are based on mortality data (hatchlings found dead) provided in Table 4.20.

The SPEARMAN's rank correlation test revealed that there is a positive correlation between hatchling disorientation and hatchling mortality per beach areas [H_1: p (two tailed) = 0.000001].

This indicates that hatchling mortality is significantly increased in disoriented hatchlings.

4.2.5 Hatchling orientation experiments in the field

Nearly all hatchlings started crawling in the experimental setups, both in the absence and presence of light cues (Figure 4.20). These hatchlings (n = 58) were included in the binomial statistics. Hatchlings that did not leave their start point (n = 4) were assessed as inactive and excluded from statistics.

Setup 1 (n=10)
Lights: -
Sounds: +
Substrate vibrations: +

Setup 2 (n=19)
Lights: -
Sounds: +
Substrate vibrations: -

Setup 3 (n=17)
Lights: +
Sounds: +
Substrate vibrations: +

Setup 4 (n=16)
Lights: +
Sounds: +
Substrate vibrations: -
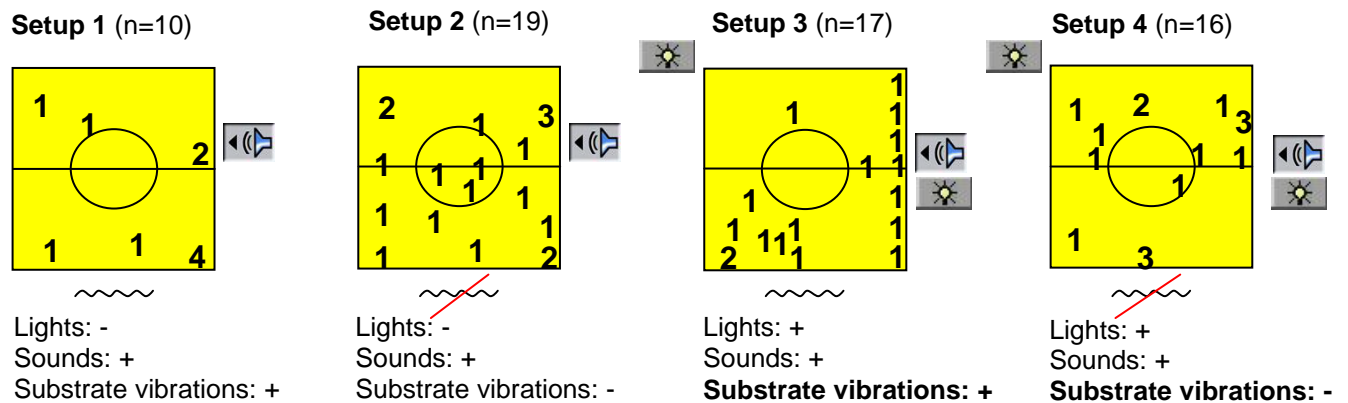
Figure 4.20: Final positions of *C. caretta* hatchlings in the experimental arena (Belek, 2005). Digits summarize number of hatchlings. Upper half: orientation landwards. Lower half: orientation seawards. Digits in the circle indicate inactive hatchlings which did not move from their start point. Symbols indicate stimuli: artificial lights (lamp symbol) and noise (speaker symbol). Note that in setup 3, including possible substrate vibrations from the shore, a high proportion of hatchlings oriented away from artificial lights (seawards). In contrast, nearly all hatchlings oriented landwards when shielded against vibrations (setup 4)

The outcomes of the FISHER'S Exact probability test comparing setups 3 and 4 reveal that in the presence of artificial lights, but shielded against substrate vibrations, hatchlings are significantly disoriented (distributed in the half of the box facing landwards).

Table 4.22: Statistical outcomes of the experimental setup with *C. caretta* in Belek

Experimental setup	Setup 1 (n=10) hatchling activity: 100%	Setup 2 (n=19) hatchling activity: 84%	Setup 3 (n=17) hatchling activity: 100%	Setup 4 (n=16) hatchling activity: 95%
Sign test	Hatchlings are equally distributed (H_1: p = 0.754)	Hatchlings are equally distributed (H_1: p = 1)	Hatchlings are equally distributed (H_1: p = 0.332)	Hatchlings are equally distributed (H_1: p = 0.118)
FISHER'S Exact probability test[1]	Hatchlings are equally distributed in Setups 1 and 2 (H_1: p = 0.701)		Hatchlings are unequally distributed in Setups 3 and 4. (**H_1: p = 0.042**).	

[1] In Setup 3 hatchlings show significant orientation seawards, whereas in Setup 4 hatchlings significantly orient landwards. In Setups 1 and 2 there is no significant difference in distribution.

4.3 Discussion

Coastal development and light pollution:

In 2005, the Natural Site Area (Section I) was affected by coastal development far less than the Tourism Development Area (Section II). This agrees with the earlier assessment made by CANBOLAT and NALBANTOGLU (2001). Based on satellite data, I demonstrated that in Section II there was a considerable increase in coastal development within the last years (Figure 4.6). This is consistent with plans for building projects and also agrees with the latest statistics, showing increasing tourist numbers in this area (Figure 4.2). Comparing coastal development and light pollution in the study area, I made an alarming finding. Though being nearly undeveloped and lacking artificial light sources set up here (low ALSD), Section I was considerably affected by sky glow and direct irradiation originating from adjacent light sources along its entire beach length (Appendix 3). This had an impact on the natural nighttime brightness in this Natural Site Area, mainly in areas close to developed plots. The flat character of the coastal zone, which did not provide any light barriers along its east-west axis, amplified the vertical light propagation along the shoreline. Sand dunes and associated vegetation in the north did not have any effect on the visibility of remote light sources here. The absence of light-blocking features is a crucial problem at nesting beaches, affecting hatchling orientation (WITHERINGTON and MARTIN, 1996). This was evidenced by disoriented hatchlings that were found in Section I, mainly in its western area, close to artificial lights. Disorientation was remarkably low in the eastern area of Section I, which seemed contradictory given increased light pollution also in this area. This is explained by the generally lower nesting density, resulting in lower hatchling output here. The low nesting density found here agrees with the thesis that females avoid nesting in areas of intensive artificial lighting and prefer darker areas (WITHERINGTON, 1992a).

Calculating the Artificial Light Source Density (ALSD) in my thesis was practical for quantifying major stationary light sources, such as Metal Halide (MH) and High Pressure Sodium (HPS) lights in Section II. However, it was not found practical for smaller light sources, including incandescent lamps, due to their multiplicity in the hotel zones. But I was able to find a negative correlation of ALSD and *C. caretta* nest densities, using the ALSD for MH and HPS lamps alone. In contrast, a positive correlation between ALSD for these models and hatchling disorientation could not be proven. As the ALSD for MH lights was nearly consistently high within the four hotel

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zones (ALSD: 5.1 to 9) compared with public areas (ALSD: 0 to 2) or Section I (ALSD: 0), the ALSD calculated here could also be used for extrapolations on ALSD for similar infrastructures, e.g. hotel zones or water sport areas at other nesting sites. Therefore it may serve for identifying areas of increased light pollution without assessing the situation in the field, based on high-resolution (15 m) ASTER satellite data showing coastal infrastructure (Figure 4.16). Hence it could be used to extrapolate sea turtle nesting densities at other sites and make prognoses on the effect that coastal development has in these areas. However, other nesting sites may have different infrastructure and lighting installations, resulting in different ALSD. Comparative studies in the Gulf of Antalya are recommended here. Nevertheless, being an exclusively quantitative tool, the ALSD takes into account neither the intensity of the light sources evaluated nor the direction of irradiation. Thus it does not reflect the effective light propagation towards undeveloped areas. Therefore in addition to the ALSD calculation I measured illuminance levels in the egg-laying zone. This revealed remarkable differences between beach areas. Natural conditions close to nocturnal ambient light (0.05 lux – 0.11 lux) were found only in the central area of Section I. These results are consistent with literature values of illuminance on the ground, ranging from 0.01 lux at quarter moon to 0.25 lux at full moon at sea level on a clear night (JANICZEK and DEYOUNG, 1987, SPUDIS, 1999). However, in Section I did not have values below 0.01 lux at new moon, which would characterize absolutely dark sea turtle nesting beaches (WITHERINGTON, 1992a). The reason for this was light pollution from adjacent areas, which had a cumulative effect and increased the overall illuminance in Section I. This clearly indicates that even the darkest area of Section I was not completely free of light pollution. In contrast, Section II was many times brighter than Section I along its total beach length, since illuminated by a variety of nearby artificial light sources. HPS lights (250 watts) along a coastal street increased overall Illuminance in the egg-laying zone by a factor of 5, and MH lights even by a factor of 24 (70 watts) to 160 (1000 watts). These lamps also had a considerable upward light flux due to poor shielding properties. It is pointed out that all values on illuminance measured here have to be considered carefully, as they may not reflect the effective situation. Strong coastal winds often changed the sky cover and visibility of celestial bodies, and consequently the illuminance on the beach (JANICZEK and DEYOUNG, 1987). Another point is that illuminance captures light spectra from 380 nm to 780 nm, representing human eye

sensitivity, but it does not measure light spectra outside this range accurately. This has to be taken into account, considering that sea turtles have a shifted visual range compared with humans, perceiving UV radiation below 380 nm, but generally being less sensitive for visual light of > 600 nm (WITHERINGTON and BJORNDAL, 1991a). Thus measuring the illuminance of light sources alone is suitable only to a limited extent to assess the effects on sea turtles and their hatchlings. Knowledge of the spectral characteristics of the light sources is crucial (see Chapter 1, Table 1.3). This could be a starting point for developing special lux meters taking into account sea turtles' visual range. Using the lux meter as a standard measuring method on the ground, my results on overall illuminance in Belek tend to confirm the increase in artificial night sky brightness in this area. Based on the light pollution maps with an effective ground resolution of 2.8 km provided by CINZANO et al. (2001a), I assessed the Antalya/Belek region as up to three times brighter than the reference value for absolutely dark sky in 1996/1997 (see Chapter 3, Figure 3.9). Due to increasing coastal development in the Belek tourism area, which is evidenced by the number of new hotels build in recent years, it is most probable that overall light pollution levels have exceeded the 1996/1997 levels over the last decade by far. This is supported by the estimation of the latter authors, indicating a 10% increase in light pollution per year in areas where no counter-measures were taken (FALCHI, pers. comm.). Thus for the Belek nesting area I assume a twofold or even greater increase in light pollution since 1996/1997, making this site more than six times brighter at the zenith compared to natural light levels today. Satellite data providing information on the wavelengths emitted by stationary lights may help to capture particular light sources in the Mediterranean that are harmful for sea turtles. Combined with regular visual inspections in the field, this could contribute to establish effective conservation methods for the future. Results of this fieldwork emphasise the need to conduct further investigations, to record and map light pollution at all Mediterranean sea turtle index nesting sites, because it is the primary cause of hatchling disorientation and increases mortality on land. When developing mitigation measures, a focus should be put on preventing light propagation from distant sources, as it also affects undeveloped or protected areas. This is also probable for Sections III and IV located east of Section II, which are Specially Protected Areas (SPA) in Belek. Further investigation in the field is needed here. MH lights contribute to high light pollution levels in Belek and are also known to be highly disruptive for hatchlings due to their

broadband properties (WITHERINGTON and MARTIN, 1996). Developing mitigation measures for this particular light source was assessed as a top priority in my thesis (see Chapter 5).

Female nesting density, hatchling disorientation and mortality:

Comparing the distribution of nests in 2005 with earlier data, I found evidence for a spatial shift in nesting in the Belek nesting area. In 1999 and 2000 slightly more nests were found in Section II than in Section I (CANBOLAT and NALBANTOGLU, 2001), but in 2005 this ratio was changed. More nests by far were found in Section I than in Section II (CANBOLAT, pers. comm., pers. observation). Under the precondition that the abiotic factors of the beach (sand structure, temperature, humidity) in Section II remained constant over the years, this confirms that coastal development and light pollution had a negative effect on nesting sea turtles. This is also seen in the statistical evaluation of the nesting density (ND) in my thesis. The ND in hotel zones (Section II) was considerably lower than the ND in Section I. Moreover in Section II the ND was higher in public areas and areas under construction than in the hotel zones (Student's t-test). In total, the ND in Section I was up to four times higher than in Section II. It was also confirmed that the nesting density was negatively correlated with the Artificial Light Source Density (ALSD), which was defined in my thesis (see SPEARMAN's rank correlation test, Table 4.7). In fact there was low ND observed in front of HPS lights and Metal Halide lights, which is consistent with the females' aversion for brightly lit beaches (WITHERINGTON, 1992a). Despite high illuminance measured in front of polychromatic Metal Halide lights, remarkably I still found some nests in close proximity to these lights. But hardly any nests were found close to HPS lights, which were contributing to the comparatively low illuminance in the egg-laying zone. It is suggested here that this is based on different times of operation. Whereas Metal Halide lights in front of hotels or construction sites were generally switched off in the late night or early morning, HPS were in use constantly until dawn to illuminate the coastal street. This is a particular source of danger for hatchlings. Females may wait offshore until the Metal Halide lights are off, but hatchlings emerging in the late evening or early night would be inescapably exposed to these bright lights. Thus the nocturnal operation of Metal Halide lights is unacceptable, as it interacts with both the nocturnal arrival of *C. caretta* females and hatchling emergences in this area (CANBOLAT, pers. comm.; pers. observation). This highlights the need for technical adjustments here. Comparative data for other highly developed beaches are

relatively sparse. SALMON et al. (1995a) conducted research on an urban *C. caretta* nesting beach in Florida, USA. The authors found that the nesting density in front of condominiums with high silhouettes was higher compared with beach zones in front of parks with lower-silhouette trees. This is remarkable, as females would be expected to choose sites in front of vegetation, which provides more natural cues for nesting. The same authors found that overall light intensities were almost equal in front of condominiums compared with sites with natural vegetation as background. They concluded that these condominiums, which were unoccupied and dark during the summer months, were attracting females by providing a higher light intensity contrast to the sky glow, as do lower profile areas with natural vegetation. The situation in Belek is different, because there is considerable light pollution in front of condominiums (hotels). In fact sea turtles significantly chose the low profile, but dark, beach areas over illuminated condominiums providing high silhouettes in Belek. In overall my results qualify data of SALMON et al. (1995a), who did not find significant correlations between the average nesting densities and light intensity at an urban beach in Florida. Based on my results on nesting densities it is also concluded that it is primarily increased light pollution levels, not the degree of coastal development itself that affects females. For comparison with the studies conducted by SALMON et al. (1995a), it would be interesting to test if females' site choice in Belek would also change in favour of higher elevation cues (hotels) if all lights could be switched off there. But this experiment is hardly feasible due to mass tourism in the area overlapping with the *C. caretta* nesting season.

In Belek, artificial lights from hotel zones make it difficult for females to find a beach area that provides enough light contrast. In Section II, direct irradiation from close light sources was omnipresent in 8 km of the total 11.3 km beach length, forcing females to nest on the remaining 3.3 km, which are public areas (see Appendix 3 c-f). As a consequence, public areas between the hotels were attracting females by providing a light intensity contrast to bright hotel zones. It is stressed here that lights shining on the beach at a right angle may be of particular risk to hatchlings. Being less visible from the sea side, these lights may not deter females to the same extent as the same light directed towards the sea in a straight angle. But hatchlings emerging on the beach would face the whole extent of illumination and become light-trapped. This was evidenced by notably high hatchling disorientation observed in the public areas of Belek (Appendix 3 c-f). This is a negative example of the

anthropogenic influence that is outmanoeuvring the females' inherent protective mechanisms. Another option for gravid females waiting offshore would be to shift to more undeveloped areas, like Section I. It remains unclear to what extent sea turtles will be able to adapt to new nesting areas, as they show a high site fidelity and preference for particular beach sections (MROSOVSKY, 1983, MILLER, 1997). This should be investigated over time by monitoring individual females' site preferences, using tag marks for identification (see Chapter 2). It is concluded here that further development of the hotel zones, and associated increase in light pollution, will probably reduce female nesting success and hatchling output, contributing to a long-term decline in the Mediterranean sea turtle populations.

My evaluation of the hatchling disorientation data for the 2005 nesting season revealed that in Section I 5.5% of all hatchlings were disoriented, whereas disorientation was notably higher in Section II, with 28.7%. This is based on the statistics, differentiating orientation either "seawards" (south) or "landwards" (north) in the circular arena (Figure 4.4). For Section II, this calculation is nearly consistent with the findings of CANBOLAT and NALBANTOGLU (2001), who indicated 33.5% hatchling disorientation in Belek in 1999 and 2000. The authors do not specify if this was for the whole Belek nesting area (Sections I, II, III and IV, Table 4.1) or for the Tourism Development Area (Section II) only. It is pointed out that for the 1999/2000 assessment these authors considered all hatchlings as disoriented that moved to the west, to the east, or to the north (three-quarters of the circular arena). But using my calculation according to Figure 4.4, this would account for about 24.5% of the disoriented hatchlings in 1999/2000, which is lower than my calculation of 28.7% (Section II) for the 2005 season, meaning an increase in disorientation. Such an increase in disorientation rates within five years would be reasonable, due to the increase in coastal development and overall light pollution in the study area (Fig. 4.6 b, Appendix 3 f). At other Mediterranean sites, IRWIN et al. (1996) indicated 39% disoriented hatchlings in Northern Cyprus, in an area affected by light pollution. PETERS and VERHOEVEN (1994) counted 63% disoriented *C. caretta* in the Göksu Delta, Turkey, near strong artificial light sources from a paper factory and a holiday village, which were not specified in detail. These numbers are alarming, considering the present high hatchling mortality at the Mediterranean beaches.

Hatchling disorientation was not increased in areas of higher ALSD (see SPEARMAN's rank correlation test, Table 4.8). At first glance this appears contrary to the

established theory of disorientation by polychromatic lights. In fact, I found extreme hatchling disorientation in front of Metal Halide lights, which are known to have a negative effect on the orientation of sea turtle hatchlings (WITHERINGTON, 1992b, WITHERINGTON and BJORNDAL, 1991b, WITHERINGTON and MARTIN, 1996). This agrees with the thesis that hatchling perception of specific wavelengths increases with light intensity (WITHERINGTON and MARTIN, 1996). So what is the reason for the negative outcomes of the SPEARMAN test here? First, the ASLD does not reflect the effective impact of light pollution, as it neither differentiates the direction in which the light source emits nor the distance of a nest from the light source, as discussed above. Second, it does not account for light propagation from adjacent areas. This is important, as hatchling disorientation is a result of all light sources that are visible on the beach. This was evident for Section I. While lacking stationary light sources here (ALSD = 0), this beach area was affected by artificial lights from adjacent areas at > 2 km range. It is stressed here that sky glow also had a negative effect on hatchlings in Section I, as it was visible from a relatively low vertical angle, at which hatchlings are believed to perceive light (WITHERINGTON and MARTIN, 1996). Research done on an urban beach in the United States also confirms the negative effect of sky glow on hatchling orientation (SALMON et al., 1995b). Third, there were a high number of smaller wattage lamps in the study area, including incandescent lamps at beach bars and hotels, which could not be included in calculations, though of course these lamps also contributed to light pollution. Finally, it is pointed out that only disorientation data that were confirmed by visual inspection of hatchling tracks were included in my statistics. There were a number of nests for which crawl tracks were not available. This included nests where the hotel staff switched their lights off shortly before hatchling emergence (n = 9). Moreover, some hatchlings were taken to the sea by the hotel staff or tourists, to prevent hatchling disorientation (n = 6). In at least one case hatchling tracks were removed by the cleaning personnel, which also led to a bias in statistics. Thus, assuming that these local protective measures were not taken, the effective hatchling disorientation would probably have been higher. Remarkably high hatchling disorientation per kilometre was found in Section I and also in undeveloped areas of Section II (public zones). First, these areas provided more nesting space, which resulted in higher nesting densities. Hence, higher nesting densities (higher hatchling output) also resulted in higher hatchling disorientation in areas affected by light pollution. Second, as outlined above, females

laid nests at sites that were not suitable for hatchlings. Undeveloped areas in Sections I and II provided more visual contrasts to brightly illuminated hotels, but hatchlings emerging here became light-trapped when moving away from contrasting dark towards brighter areas. This was particularly evident for the public areas of Section II (see Appendix 3). In contrast, hatchling disorientation was low in front of HPS lights in the east of public area 3. As outlined before, this was due to the constant working times of these lamps, resulting in low nesting densities. Moreover, this area was highly frequented by humans and vehicles (noise pollution), which probably also deterred females. Therefore the outcomes on hatchling disorientation in my thesis must be considered minimum values. Effective disorientation rates are believed to exceed these values, especially close to hotel zones. With regards to light pollution it is concluded here that the entire Section II and also the peripheries of Section I are today unsuitable for sea turtle nesting. Undeveloped or public areas between hotels hold a particular risk for hatchlings due to light-trapping.

Hatchling emergences in Belek were not dependent on the moon phase. This is the general state of knowledge (MROSOVSKY and CARR, 1967, SALMON and WITHERINGTON, 1995). I observed a highly significant relationship between the frequency of hatchling disorientation and the decrease in ambient light levels (new moon) in Section II (SPEARMAN's rank correlation test). This area was affected by high-level light pollution from nearby sources. These findings are consistent with data for a nesting beach in Northern Cyprus, which was also affected by light pollution (IRWIN et al., 1996). My results also agree with findings made at nesting beaches in Florida, USA. Under constant light pollution levels, *Caretta caretta* hatchlings showed increased disorientation on new-moon compared to full-moon nights (SALMON and WITHERINGTON, 1995). These authors suggested that background illumination from the moon, not the attraction to the moon itself, restored sea-finding orientation. This is consistent with the theory that hatchlings are not exclusively oriented towards the brightest source, but move away from dark silhouettes (WITHERINGTON, 1992b, SALMON and WYNEKEN, 1994). Based on my outcomes, I confirm a combined mechanism of hatchling orientation towards the brightest source and away from dark shapes. At natural beaches the darkest direction is landwards. But contrasts are changed if this direction appears brighter due to artificial lights (Section II). An increase in ambient light, such as the full moon, restores these contrasts to a certain extent, making the sea relatively brighter again. In Section I, disoriented hatchlings

were distributed equally over the lunar cycle. This is contrary to my findings made for Section II. SALMON and WITHERINGTON (1995) suspected a reciprocal relationship between the intensity of a disrupting light source and the amount of ambient light required to restore sea-finding. Thus, a full moon should have had an even stronger correcting affect on hatchlings in Section I, which was affected less by light pollution compared with Section II. What is the reason for this discrepancy? My findings for Section I would agree with the hypothesis that it is primarily "anisotropic", that means directed light, which has a negative effect on animals' orientation (VERHEIJEN, 1958, 1982, 1985) and which is mitigated by increasing ambient light levels. Section I was generally affected by direct irradiation ("anisotropic light"), but was also considerably exposed to sky glow, which has more "isotropic" properties as it originates from different sources and directions. This may result in a minor mitigation effect of moonlight in this area. Second, Section I did not display any high shapes or elevation gradients, due to its naturally low beach profile along the Besgöz River. This lack of elevational cues may also have reduced the sea-finding ability of hatchlings. This was observed in the field; some hatchling tracks were leading directly into the Besgöz River, located opposite to the seaward side. In contrast, Section II provided enough elevational cues (condominiums). Results for Section I also would agree with the findings of PETERS and VERHOEVEN (1994), who conducted research in the Göksu Delta, Turkey, in two areas of different level coastal development and light pollution. In the undeveloped area, hatchling disorientation was constant over the lunar cycle. The authors do not specify if disorientation was also constant for the developed area, located in front of a holiday village and adjacent to a paper factory, which were causing light pollution. My findings confirm the need to reduce high levels of direct irradiation in Belek, which results in multiple illuminance on the beach compared with natural ambient light conditions. It is difficult to prescribe a threshold level for illuminance at sea turtle nesting beaches, since ideally we are aiming for the total reduction of artificial lights here (WITHERINGTON and MARTIN, 1996). But it is stressed that a slight change in ambient light levels can affect hatchling orientation significantly, especially on dark nights. Therefore it is strongly recommended that the illuminance levels near sea turtle nests should be reduced to < 0.25 lux, which is equal to the highest level of ambient light at full moon.

Results show that hatchling mortality, defined as hatchlings found dead on the beach, is higher in Section I than in Section II. Another finding is that in Section II

mortality rates are considerably lower than in Section I. This is remarkable, as mortality was correlated with higher disorientation rates (SPEARMAN's rank correlation test). As described above, in Section II the effect of human interaction on disoriented hatchlings was not negligible. Tourists or hotel staff picked up disoriented hatchlings, before these died from exhaustion or dehydration. Moreover a number of dead hatchlings were removed from the beach by the hotel cleaning staff and could not be counted during monitoring patrols. Another percentage of dead hatchlings was probably not found, as they had moved further landwards and perished under the attracting lights or coastal vegetation. In Section I, which was remote from the hotel zones in Section II, human help in reducing mortality was sporadic. By contrast, there was a higher chance of interaction with predators on this natural site, namely stray dogs near human settlements, evident in the west of Section I, or ghost crabs (*Ocypode cursor*) on the entire beach length. Whereas dogs fed on eggs or killed hatchlings, ghost crabs were natural predators on hatchlings. In 1999 and 2000 this accounted for total of 36.9 percent nest damage by predators in Belek (CANBOLAT and NALBANTOGLU, 2001). This is an extremely high predation rate at beaches where there is no human activity (CANBOLAT, 2001). It is suggested here that the assessment of mortality, which is based on the number of hatchlings found dead on the beach, does not reflect the effective mortality by far. In this context it is pointed out that there is no information available on hatchling mortality at sea in this area. Even if not causing mortality on land in every case, artificial light sources may indirectly cause mortality by exhausting hatchlings on their unnaturally extended crawls on the beach. This would reduce hatchlings' fitness, making them more vulnerable for predators offshore or heavy sea conditions and making it more difficult for them to reach their feeding and resting habitats. Therefore it must be assumed that the effective mortality rate in areas affected by light pollution is much higher. Further research on potential "delayed" effects of light pollution is recommended here.

Ambient sounds and noise pollution:

Breaking waves basically contribute to ambient sounds in coastal zones (WILSON, 1998, LOEWEN and FARELL, 1998, PRONI, 1998). I conducted measurements on surf sounds (breaking waves) when no other major sound source but wind was present on the beach. Based on comparisons with literature data it is suggested here that the outcomes of my measurements on surf sounds are representative. The sounds of a

1-m surf were clearly detectable up to 35 m from the shore. This is the maximum distance at which sea turtles lay their nests in this area (CANBOLAT and NALBANTOGLU, 2001). Though I did not conduct measurements on substrate vibrations with an accelerometer, it is suggested here that at given distance, in contrast to aerial sounds, seaborne vibrations may be partly dampened by the beach sand. My measurement revealed that at 20 m, which is the normal distance of nests from the shore, the surf still has clearly detectable sound levels. Breaking waves, which I measured here, had sound levels of up to 71 dB (SPL), with clear peaks in the low frequencies at < 500 Hz. These results are within the sound-level range of surf sounds known from literature (WILSON et al., 1998). Surf sounds have broadband spectrum characteristics. Measured sound level and frequencies also strongly depended on the phase of the breaking wave. Breakers in the surf are known to emit up to 2 KHz (KOLAINI, 1998), but with clear peaks in the very low frequency spectrum around 10-20 Hz, (KOLAINI, 1998, LOEWEN and FARELL, 1998, WILSON, 1998). This is infrasound, which my Audio Analyser was not able to capture. Other literature data characterize the near shore acoustic spectrum by two broadband peaks, in the low-frequency range from 100 to 500 Hz and in the mid-range from 1 to 7 kHz (MELVILLE et al., 1996). I found consistency in the low-frequency spectrum when increasing the measuring distance to the sound source from 5 m to 10 m, 15 m, 20 m, and 25 m. This agrees with the physical properties of sound propagation in air and water. In both media, low-frequency sounds are absorbed far less than high frequencies. This is dependent on the viscosity of the medium, which changes with temperature. My measurements on land are also consistent with data on wave sounds obtained underwater with hydrophones, indicating frequency ranges of 50 to 1000 Hz (WILSON, 1998).

Sensitivity to aerial sounds in sea turtles is low compared with freshwater turtles (PATTERSON, 1966). The latter species lack the thick tympanum of sea turtles, which dampens sounds (LENHARDT et al., 1985). But newer data on sea turtle acoustic perception ability qualify previous findings, which supported hearing in water rather than on land (LENHARDT, 1982, LENHARDT et al., 1983, LENHARDT et al., 1985). Sea turtles are capable of perceiving low-frequency sounds in different media (RIDGWAY et al., 1969, BARTOL et al., 1999, BARTOL and KETTEN, 2006). LENHARDT (1994) found *C. mydas* to be sensitive to aerial low-frequency sounds at 50 to 100 db re 20 μPa, whereas threshold levels for sounds underwater were higher at 150-200 dB re 1 μPa.

Preliminary findings on the inner ear morphology support air-conducted hearing, in particular in sea turtle hatchlings (LENHARDT, 2005). Based on my measurements of surf sounds, emitting up to 71 dB (SPL) at 20 m distance, I suggest that the low-frequency component of this sound is clearly within the hearing range of sea turtle hatchlings in the egg-laying zone. I found a remarkable conformity of the *C. mydas* hearing sensitivity curve (RIDGWAY et al., 1969) and the frequency curve of surf sounds between 50 and 400 Hz (Figure 4.18). These findings also agree with preliminary data of NUNNY et al. (2005) supporting a match between the tonal emissions of the surf zone and the range of turtle hearing. I conclude that aerial low-frequency noise pollution at high sound levels is also within the hearing range of sea turtles. Overall noise pollution, caused by discotheques and traffic along the coastal street, was high in the Tourism Development Area (Section II) in Belek. This is a common problem for sea turtle nesting sites, which are affected by mass tourism (MARGARITOULIS, 1990, VENIZELOS, 2001). As both surf sounds and the anthropogenic noise sources measured have broadband properties, it is suggested here that noise may mask natural wave sounds when emitted at high intensities close to sea turtle nests. The Masking effect also depends on the wave height, since that is the determining factor for the sound level of the surf (WILSON, 1998, LOEWEN and FARELL, 1998, PRONI, 1998). It is suggested here that breaking waves at > 1 m, resulting in high sound levels, may dominate the ambient sounds in the egg-laying zone and probably mask noise. However, with a calm sea there may be the risk that noise would mask surf sounds.

In this chapter I demonstrated that surf sounds are within the hearing range of sea turtles. But the function of low-frequency sound perception in sea turtles remains unclear. It could be an adaptation to a coastal habitat, analogous to sea turtles' adaptation to the shorter visible wavelengths in a marine environment (GRANDA and O'SHEA, 1972). This would support the hypothesis that the low-frequency sound spectrum of the natal beach may serve sea turtles as one of the cues in nesting return (LENHARDT et al., 1983). As sea turtles do not return to their nesting beach until reaching maturity, they need to imprint on these sounds in the nest or as hatchlings. This is only possible, when crawling on the beach, or immediately after entering the sea. At a Turkish nesting site, PETERS and VERHOEVEN (1994) observed disoriented *C. caretta* hatchlings lost in dunes finally reaching the sea. This correction of heading direction in disoriented hatchlings may be by chance, elicited by a change in visual

cues, or by alternative orientation cues such as beach slope (VAN RHIJN, 1979). In Belek I made similar observations at a low profile beach with minimal beach slope and illuminated background. *C. caretta* hatchlings first oriented towards artificial lights close to the nest, but then turned back to the sea, which was the darker direction (Figure 4.4 A). Surf sounds were clearly audible while hatchlings showed a change in heading direction. According to today's state of knowledge, this behaviour would be elicited by changing brightness and shade cues while crawling (SALMON and WYNEKEN, 1994, WITHERINGTON and MARTIN, 1996). May this behaviour also be caused by the interaction of acoustic cues with artificial lights? A decrease in surf sound intensity may be detectable by hatchlings when orienting landwards and could elicit a change in direction. By contrast, I found that hatchlings were not able to find the sea under similar sky conditions and surf sound levels, but in the presence of stronger light sources, radiating from a smaller distance (Figure 4.4 B). Thus it is assumed here that acoustic cues may only have a correcting effect on hatchlings if the intensity of the attracting lights is comparatively low. For testing interactions between artificial lights and sounds, I conducted behavioural experiments with a small sample of *C. caretta* hatchlings. All hatchlings failed to orient in complete darkness and showed equal distribution in the experimental box. Probably this was due to a total lack of visual orientation cues, which prevented hatchlings from finding the sea (CARR and OGREN, 1960, VAN RHIJN, 1979). In contrast, hatchlings' aversion to or preference for airborne sounds in the absence of visual cues could not be proven. In the presence of lights, hatchlings also failed to orient seawards significantly (Sign test, see Table 4.22). This was caused by artificial lights shining from the east and northwest. These results are consistent with the established theory that hatchlings are dependent on visual cues for orientation. By contrast, I found differences in hatchling orientation when comparing two setups in the presence of lights. When shielded against substrate vibrations, hatchlings were significantly oriented landwards. But unshielded hatchlings oriented seawards (FISHER'S Exact test). Based on the outcome of this test and literature data confirming sea turtle acoustic perception ability, I suggested that in addition to visual cues hatchlings might use auditory cues, presumably substrate vibrations, for seaward orientation. The results of these explorative field experiments needed to be verified with higher sample sizes and under controlled experimental conditions. This will be discussed in Chapter 5.

5 Xcacel, Mexico: Testing artificial lights and sounds on sea turtle hatchlings

5.0 Introduction

To date, behavioural studies on the acoustic perception ability of sea turtles and hatchlings in particular are sparse. Whereas the negative impact of artificial lights on hatchlings is proven in experiment and established as the main factor of hatchling disorientation (WITHERINGTON, 1991, WITHERINGTON and BJORNDAL, 1991b, WITHERINGTON, 1992b, SALMON and WITHERINGTON, 1995, SALMON et al., 1995b), a possible effect of sounds on the behaviour of hatchlings on land has not been adequately investigated and is still speculative. MANGIAMELE and LOHMANN (2005) investigated the role of wave sounds as an orientation cue for *C. caretta* hatchlings. They placed hatchlings in the centre of a runway in complete darkness and exposed them to wave sounds emanating from a speaker at one end of the runway. The authors found that hatchlings did not significantly orient to aerial wave sounds in their experimental setup. At present, this is the only preliminary behavioural data available on hatchlings tested for sounds (see Chapter 1). NUNNY et al. (2005) investigated the physical properties of surf sounds and found a clear match between tonal emissions of the surf zone and the range of turtle hearing. The authors suggested that sea turtles could be using acoustic cues to control their nesting behaviour. These findings also support the hypothesis that surf sound may serve sea turtles as a cue when returning to their nesting beaches (LENHARDT et al, 1983). LENHARDT (2005) presented preliminary evidence that the ears of sea turtle hatchlings had fully-developed hair cells, but less limbic hair cells than adults. Limbic hair cells were assessed as being primarily receptors for vibrations and bone-conducted hearing, whereas basilar membrane hair cells more likely enabled air-conducted hearing. The author concluded that the developmental form of hearing gradually shifts from primarily air conduction in the hatchling to bone conduction in the adults. If hatchlings are capable of perceiving low-frequency aerial sounds or vibrations on land the question remains, what is this ability for? In Belek, I recorded frequency spectra and sound pressure levels of surf sounds. Comparing my measurements with electrophysiological data from RIDGWAY et al. (1969), it was demonstrated that low-frequency surf sounds are within the hearing range of *C. mydas* juveniles (Chapter 4). In fact, the surf is the predominant sound source at coastal zones (WILSON, 1998,

LOEWEN and FARELL, 1998), which is the natural sea turtle nesting habitat. Increased sensitivity to aerial sounds in sea turtle early life stages, as suggested by LENHARDT (2005), would provide the physical background for hatchling detection of surf sounds while on land. In outdoor experiment I found that *C. caretta* hatchlings exposed to artificial lights, showed significantly reduced seaward orientation when shielded against substrate vibrations from the surf. Though this first test run was made with a limited number of animals, I concluded that substrate vibrations caused by the surf may be perceived by sea turtle hatchlings, and possibly even serve as an orientation cue with a correcting effect on sea-finding, but this needed verification. My first research approach in Mexico was to find out if *C. caretta* and *C. mydas* hatchlings showed an orientation towards wave/surf sounds in total darkness. This experiment was comparable to the study conducted by MANGIAMELE and LOHMANN (2005), but testing two species using different test sounds and also two different mechanisms of sound propagation. For this, sound conditions on a natural beach were simulated in an indoor experiment, using a setup providing aerial and vibratory wave/surf sound stimuli from disc. Additionally I tested artificial low-frequency sounds for simulating noise pollution, which I identified as interacting with wave/surf sounds in the egg-laying zone of Belek (Chapter 4). This latter setup aimed to investigate possible adverse effects of noise on hatchling behaviour. In a second test run, light stimuli were tested on hatchlings in the absence of acoustic stimuli. Polychromatic light sources in Belek were identified as being highly disruptive, causing disorientation in hatchlings. The objective was to find out in experiment if commercial dichroic filters, cutting out a defined proportion of a polychromatic lamp's spectral emission, significantly reduced disorientation in *C. caretta* and *C. mydas* hatchlings. In a third test run, I tested wave/surf sounds and light stimuli on hatchling orientation behaviour simultaneously. In this combined setup it was investigated if wave/surf sounds had a mitigating effect on hatchlings disoriented by light stimuli. These tests were conducted to verify my preliminary results on hatchling orientation in the presence of multiple stimuli in Belek (Chapter 4). All hatchlings previously tested indoors under experimental conditions were also tested outdoors in a circular test arena. The aim was to find out if the hatchlings, which did not move in the indoor setup, also showed inactivity under natural conditions. In addition, I tested whether the experimental treatments in the indoor setup affected the later sea-finding ability of hatchlings. This would be relevant for the planning of conservation measures.

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5.1 Methods

Site selection and spatial characteristics of the nesting beach

All experiments were conducted at the Sea Turtle Camp Xcacel (20.339°N, 87.348°E), which is located on the Caribbean coast of Mexico (Quintana Roo state), 43 km south of Playa del Carmen and 17.5 km north of Tulum. Xcacel is one of the most important index nesting sites in Mexico, holding both *C. caretta* and *C. mydas* nesting populations on one beach (MARQUEZ, 1990, ZURITA et al., 1993, ENCALADA et al., 1999). These are the same species that also nest in the Mediterranean. For *C. caretta*, Xcacel and adjacent beaches on the Yucatan Peninsula have global Top ten nesting site status, and are also globally important for *C. mydas* (see Chapter 2). In contrast to the majority of the Mediterranean sites, Xcacel has a comparatively low degree of light pollution (Figure 5.1), making this site more suitable for testing hatchling orientation under natural conditions. The camp (Figure 5.2) is maintained by "Flora, Fauna y Cultura de Mexico A.C.", a Non-Governmental Organization (NGO) monitoring the *C. caretta* and *C. mydas* nesting populations from the beginning of April until the end of October, which is the nesting season of both species in this area. A sea turtle hatchery, an outdoor facility for secure incubation of sea turtle eggs, is located 20 m from the camp (Figure 5.3). Nests found in close proximity to the tidal zone, or in areas frequented by human visitors, were usually relocated to the hatchery by the sea turtle monitoring staff. This guaranteed a constantly large number of hatchlings of both species and also facilitated a quick transfer to the camp, where I conducted my indoor experiments. I started fieldwork at the end of August, which is the peak of hatchling emergence in Xcacel. This period was suitable for obtaining a reasonable number of hatchlings of both species for statistical evaluation. *C. caretta* and *C. mydas* hatchlings were taken from the hatchery in the late afternoon or evening before being tested. They were extracted from their underground nests before the main emergence event was expected. This is $\pm$ 60 days after deposition of the nest, detectable by a depression in the sand surface above the egg chamber. Hatchlings extracted were transported to the nearby camp and kept in covered buckets until nightfall to imitate conditions in the nest and to leave them dark-adapted for the experiments. All test animals were used in the same night for the experiments within 6 hours after excavation. This ensured the naturally increased crawling activity of the test animals (frenzy). Information on hatchlings' inborn "frenzy phase" is also given in Chapter 2.0 of this thesis.

Sea turtle camp Xcacel, Mexico

Figure 5.1: Location of Playa Xcacel, Quintana Roo state, on the Caribbean coast of Mexico. Note comparably low light pollution (category 2, grey) at this site (see Chapter 3, Table 3.1).
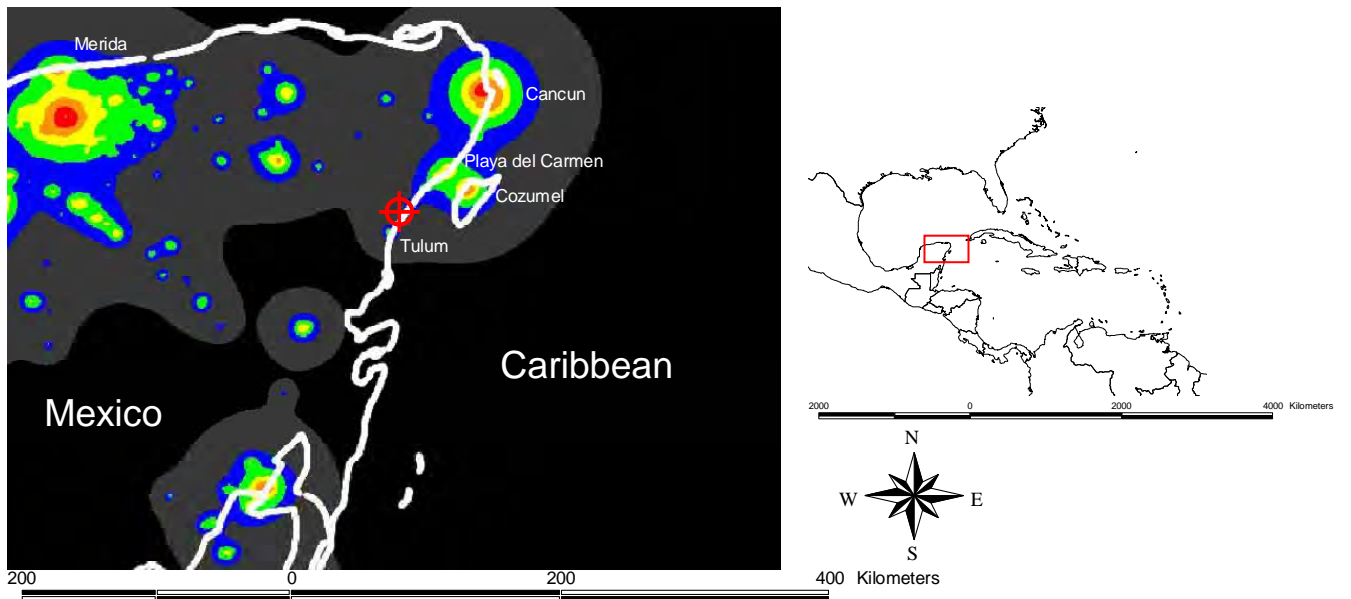

Figure 5.2: Sea turtle monitoring camp, Xcacel.

Figure 5.3: Sea turtle hatchery, Xcacel

Figure 5.4 a: *Caretta caretta* hatchlings
from the hatchery used for the experiments

Figure 5.4 b: *Chelonia mydas* hatchlings
from the hatchery used for the experiments

The experiments were conducted in the night (after dark) between 9 pm and 5 am. This is the time when the majority of hatchlings emerge from their underground nests and head to the sea (WITHERINGTON et al., 1990).

Two different experimental setups were used successively to test hatchling behaviour:

Indoor setup (Setup 1): Orientation behaviour of *C. caretta* and *C. mydas* hatchlings was tested indoors in a V-shaped two-choice box in the presence of adjustable light and sound stimuli (Figure 5.5).

Outdoor setup (Setup 2): Crawling behaviour was tested in a circular arena under natural outdoor conditions (Figure 5.9).

All experiments were conducted according to international conservation laws; none of the hatchlings was physically harmed in the course of my behavioural experiments.

5.1.1 Two-choice box experiments (Setup 1)

The experimental two-choice box was constructed V-shaped, allowing hatchlings' orientation in two directions, towards or away from the stimulus (Figure 5.5). This construction was chosen over a T-shaped one for two reasons. First, it was expected that a V-shaped box would provide more definite results, mainly for stimuli eliciting aversion in hatchlings. In a V-shaped box a hatchling may have the chance to move away from a stimulus to the opposite direction of the V-maze. In contrast, in a T-shaped box it would not have an alternative but would stay in its start position to keep away from the stimulus, as crawling inevitably would lead it closer to it. Second, a V-shaped construction allowed better comparison of my findings with similar studies. A similar setup was also used by WITHERINGTON and BJORNDAL (1991a). These authors tested *C. caretta* and *C. mydas* hatchlings for a light source of constant colour at 520 nm over an adjustable source, using narrow band filters.

85 cm

B

W

A

W

W

C

Protection

25 cm

Low frequency speaker

Tungsten Halogen Lamp

A

B

C

W

W

W
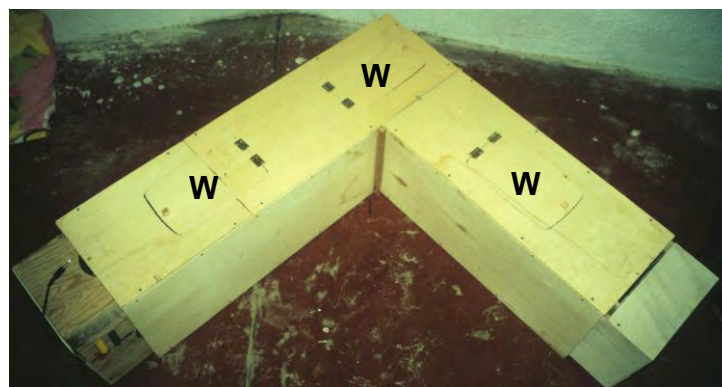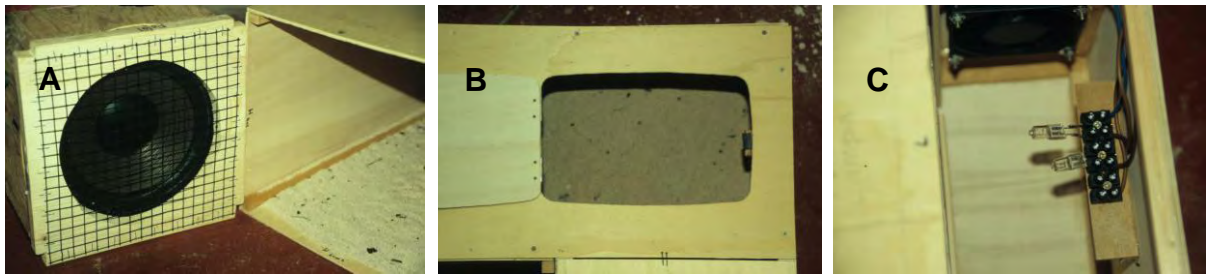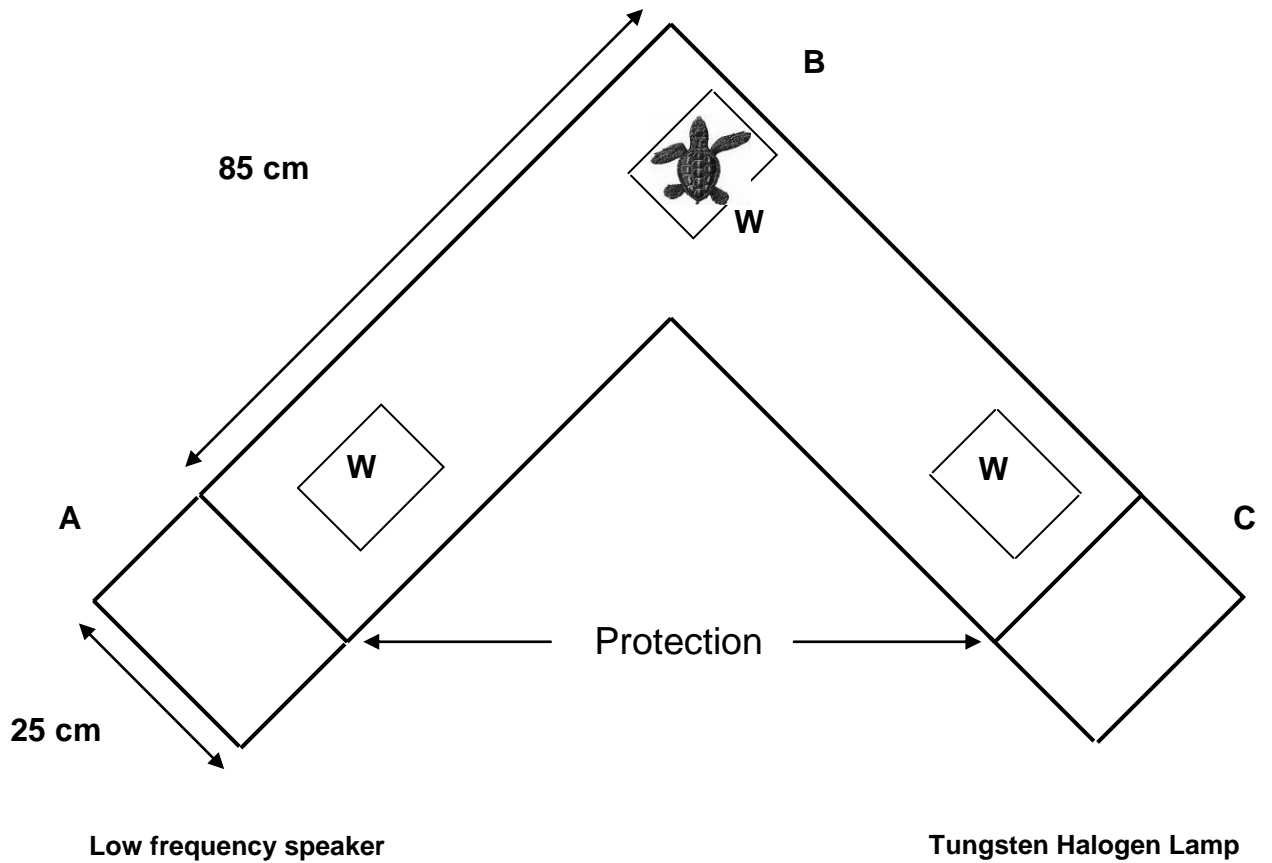
Figure 5.5: Two-choice box used for Setup 1, allowing hatchlings to move towards or away from a stimulus. Sound and light stimuli could be operated independently. W (n=3) indicates three upper windows for operation of the box. A: low-frequency speaker, B: vertex window (hatchling starting point), C: Tungsten Halogen lamp with replacement bulb and fan for cooling. The speaker was protected against hatchling crawls by wire netting. The lamp was protected by acrylic glass and/or filters placed in front of the light source.

The experimental box was mounted indoors in the camp to avoid destruction by heavy rainfall, strong winds and human/animal interaction on the beach. The experimental site was deserted when conducting the experiments, as this was the time of sea turtle monitoring nightshifts at the beach. Furthermore this location was completely dark, to let hatchlings become dark-adapted. Wave/surf sounds from the shore located 37 m from the camp were nearly undetectable from the position of the experimental site, since it was shielded by the camp's walls and windows. The two-choice box, with each identical arm 85 cm in length and 25 cm in width and height, was placed on the concrete ground of the camp, with both arms facing landwards. The box was constructed to be light-impermeable with 3 upper opaque windows (Figures 5.5). One window was built near the vertex to put the hatchlings in, and two at either end of the V-box to monitor the position of the hatchlings, and to take them out of the box after the treatment. The bottom of the box was filled with beach sand to imitate natural conditions for hatchling crawling. As far as feasible (availability), hatchlings tested in one treatment group were taken from different nests to ensure genetic diversity. All hatchlings were tested individually and a hatchling was only used for one treatment. For operating the experimental box in the dark a low intensity red light LED torch was used, which was considered to minimally affect sea turtle hatchlings. Immediately before testing, the hatchlings were taken out of their covered buckets and checked physically for activity. Individuals that did not show activity by moving head or flippers were rejected and excluded from the experiments. Active hatchlings were placed in the box individually, with their heads pointing toward the vertex of the box. The upper window was closed. The stimulus was turned on immediately after placing hatchlings in the box. Start time of the treatment was recorded. The position of the hatchling in the box was controlled every minute through the windows at the top of the experimental box. Hatchlings tested for lights could be easily tracked, whereas the position of hatchlings tested for sounds in the dark was checked with the red LED torch.

Three general patterns of hatchling orientation behaviour, attraction, aversion, or indifference to a light and/or sound stimulus, were observed. Hatchlings crawling towards the stimulus were assessed as "attracted" by the stimulus. Hatchlings that crawled in the opposite direction showed "aversion" to it. Hatchlings that did not move and remained at the starting point after 5 minutes were assessed as "indifferent" to a specific stimulus. The behavioural patterns observed are described

in detail in Chapter 5.2. After reaching one end of the two-choice box, or indifference within 5 minutes, whichever occurred first, a hatchling's final position was recorded and the treatment for this individual hatchling finished. The hatchling was removed from the experimental box and placed in a dark bucket together with other individuals of the same treatment group which had already finished treatment. The hatchlings were kept in this bucket for a maximum of 75 minutes, depending on the time flow of the indoor experiment. Usually the first hatchlings used in a treatment group remained in the bucket for over 60 minutes while subsequent hatchlings were tested. In contrast, the last hatchling tested in a group remained in the bucket for no longer than one minute. This was equal to the time required to take the hatchlings to the beach (37 m) for conducting the outdoor experiments (Setup 2). Consequently, hatchlings tested in presence of lights were in different states of dark-adaptation when tested in Setup 2, which is discussed in Chapter 5.4.

The schedule of events and the corresponding experimental time frames per hatchling are summarized below. Time frames in the lower row indicate the inter-experimental timelines (dark-adaptation phase).

Hatchery ➔ Setup 1 (max 5 min) ➔ Setup 2 (max 10 min) ➔ hatchling release

   (maximum 6 h)   (1-75 min)
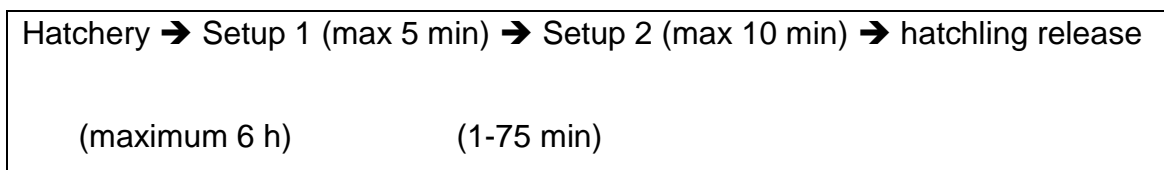
In total, 1219 hatchlings, 604 *C. caretta* and 615 *C. mydas*, were used for the indoor experiments, according to Table 5.1.

Table 5.1: Main test groups used in indoor experiments (Setup 1)

Main test groups[1]	*Caretta caretta*	*Chelonia mydas*
Control group, <u>no lights/sounds</u>	16 hatchlings	15 hatchlings
Tested for <u>sounds only</u> (Treatment groups: 11 per species)	171 hatchlings	176 hatchlings
Tested for <u>lights only</u> (Treatment groups: 12 per species)	195 hatchlings	198 hatchlings
Tested for <u>sounds and lights</u> combined (Treatment groups: 14 per species)	222 hatchlings	226 hatchlings

[1] The treatment groups are specified in Tables 5.2, 5.3 and 5.4.

Control group

A control group of both species was tested without any stimulus (no lights, no sounds) to assess hatchlings' potential preference for either the left or the right side of the two-choice box. This test was done to exclude bias based on hatchlings' possible preference for one side of the box (WITHERINGTON and BJORNDAL, 1991a).

Experimental sounds testing

The wooden construction of the two-choice box was chosen to transmit airborne sounds as well as vibrations coming from the low-frequency speaker, model Pioneer B2 1110 (8") fixed to the right end of the box (Figures 5.5). The speaker was connected to a 12-volt power supply, which was interconnected to a 110-volt AC to 12-volt DC transformer. An electric supply of 110 volts was taken from the camp. A portable disc player of model Tevion MD40983 was connected to the speaker for playing different test sounds from CD (Table 5.2). The test sounds included recordings of natural wave/surf sounds, noise (fireworks, street traffic) and artificial sounds (400 Hz, 1000 Hz tones, Pink Noise). Sound levels could be adapted through a trigger on the disc player. To measure the effective sound levels that were applied to the hatchlings, the measuring microphone of the Audio Analyser was placed in the box at the starting point of the hatchlings. Three repeated measurements of unweighted sound pressure levels (dB SPL) were made per test sound, and the average value noted. Frequency curves of test sounds were also recorded (Figures 5.6 a-f). Vibrations emanating from the speaker were detectable by touching the box.

Table 5.2: Treatment groups tested for experimental sounds

Sound[1]	Low SPL treatment (dB)	High SPL treatment (dB)	Frequency (f) in Hz
400 Hz	Group 1 (40.8)	Group 8 (50.6)	400
1000 Hz	Group 2 (45.6)	Group 9 (55.7)	1000
Wave sound	Group 3 (44.0)	Group 10 (55.4)	Broadband
Surf sound	Group 4 (44.6)	Group 11 (55.1)	Broadband
Pink Noise	Group 5 (41.6)	Not done	Broadband
Fireworks	Group 6 (55.4)	Not done	Broadband
Street traffic	Group 7 (47.2)	Not done	Broadband

[1] In total 11 sound-treatment groups per species were tested. Unweighted sound pressure levels (SPL) were measured (0 db SPL re 20 µPa). Average values were measured with the HMB-TEC A316 Analyser for 1-2 minutes per test sound (Figures 5.6 a-f), depending on the duration of the sound track on the test CD.

Rationale for test sounds used in my experiments:

<u>**400 Hz, 1000 Hz** test tones (Figures 5.6 a, b)</u>:

I tested whether hatchlings reacted differently to artificial test sounds. Sea turtles have highest perception sensitivity at 400 Hz, whereas 1000 Hz seems to be the upper perception limit.
A 400-Hz test tone at 39 dB (SPL) tested by RIDGWAY et al. (1969) was perceivable by juvenile *C. mydas* (see Chapter 4, Table 4.5). The sound levels of 40.8 dB (low SPL) and 50.6 dB (high SPL) used in my experiments (Table 5.2) are equal and higher to the reference value of 39 dB(SPL). Hence, sounds used here are expected to be within the hearing range of my test animals and may elicit a behavioural response.
C. mydas juveniles showed reduced sensitivity to a 1000-Hz test tone compared with the 400-Hz tone (RIDGWAY et al., 1969). The minimum sound pressure required to stimulate his juveniles was 79 dB (SPL) (see Chapter 4, Table 4.5). Sound pressures of 45.6 dB (low SPL) and 55.7 dB (high SPL) used in my experiments (Table 5.2) were below this reference level. Hence it was expected that this 1000-Hz test sound was out of the hearing range of my test animals and did not have any effect on hatchlings in the experimental Setup 1.

<u>**Wave/surf sounds** recordings (Figures 5.6 c, d)</u>:

I tested whether hatchlings were attracted by wave/surf sounds from disc, or showed any other behavioural response. Wave and surf sounds were tested at comparable sound levels (Table 5.2), which I measured in Turkey in the egg-laying zone (Chapter 4, Table 4.15). Previously, I demonstrated that the low frequency spectrum of the surf sounds between 50 Hz and 700 Hz is within the hearing range of juvenile sea turtles (Chapter 4, Figure 4.18). Therefore, surf sounds tested in my experimental Setup 1 were also expected to be within the hearing range of hatchlings.

<u>**Artificial sounds (noise)** (Figures 5.6 e, f)</u>:

I tested whether hatchlings showed any behavioural response to noise pollution. Traffic sound and fireworks tested here represent real noise sources, which were identified and measured in Turkey in the egg-laying zone (see Chapter 4, Table 4.5). These sounds were tested at comparable sound levels in experimental Setup 1 (Table 5.2). Additionally, Pink Noise was tested for control. This sound has definite peaks in the low-frequency range with a continuous decline to the higher frequencies. Therefore this sound is also expected to cover the hearing range of hatchlings.
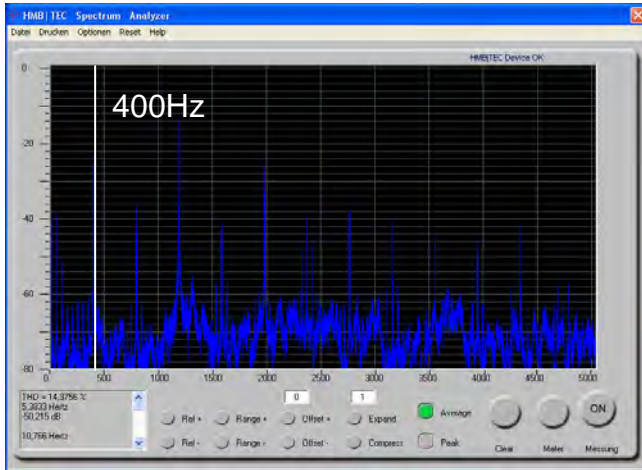
Figure 5.6 a: **400 Hz tone** (average peaks)
X: frequency (Hz), Y: output voltage (dBV)

Figure 5.6 b: **1kHz** tone (average peaks)
X: frequency (Hz), Y: output voltage (dBV)

Figure 5.6 c: **Wave sound** (average peaks)
X: frequency (Hz), Y: output voltage (dBV)

Figure 5.6 d: **Surf sound** (average peaks)
X: frequency (Hz), Y: output voltage (dBV)

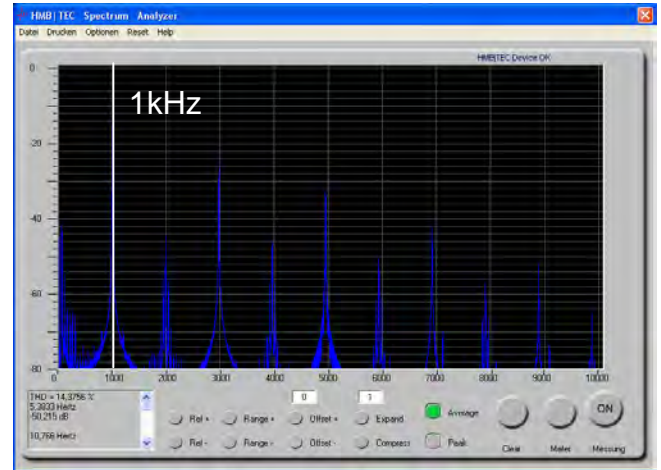Figure 5.6 e: **Pink Noise** (average peaks)
X: frequency (Hz), Y: output voltage (dBV)

Figure 5.6 f: **Street traffic sound** (average peaks)
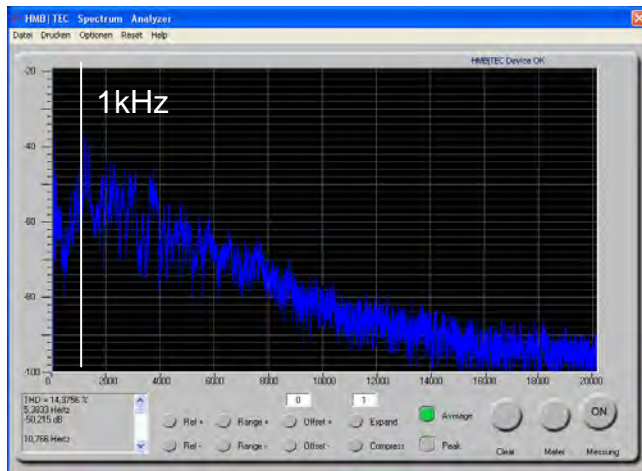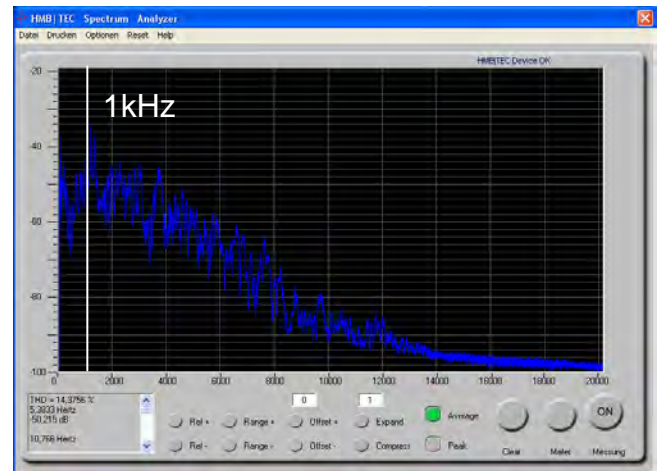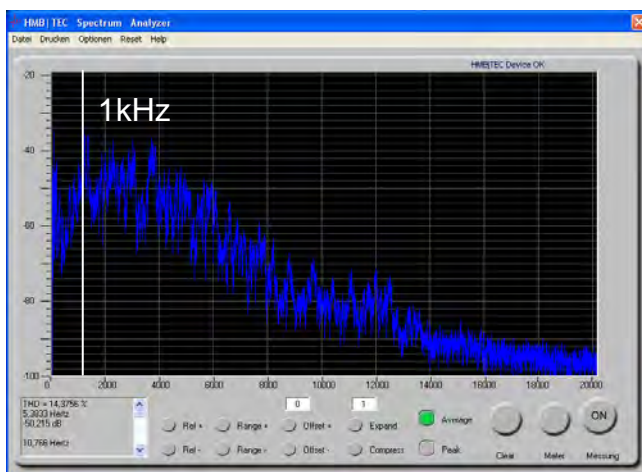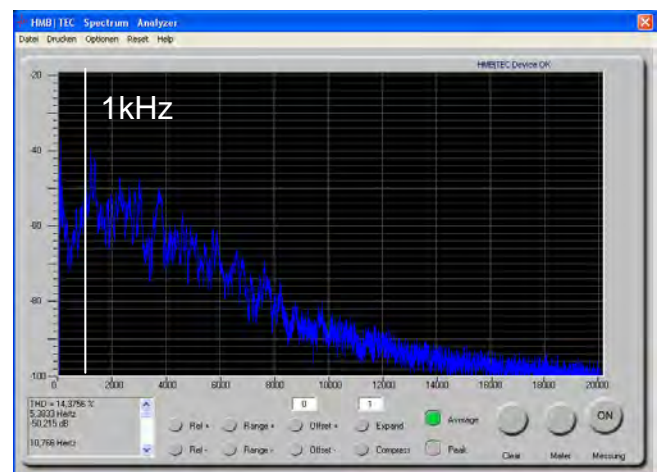X: frequency (Hz), Y: output voltage (dBV)

Figures 5.6 a-e: Note that all test sounds used have distinct peaks in the low frequency range, which is within the audible range of sea turtles (RIDGWAY et al., 1969, BARTOL and KETTEN, 1999).

Experimental lights testing

Dichroic filters were kindly provided for testing in this thesis by Dr. Antonio TERSALVI[23]. These were used with a commercial Tungsten Halogen lamp (10 watts). This lamp has polychromatic characteristics (Figure 5.8 a) with a peak at 600 nm, similar to the Metal Halide lights (see Figure 3.2 b) that were identified as a major source of light pollution in close proximity to sea turtle nests in Belek (Chapter 4). The Tungsten Halogen lamp was fixed at the left end of the experimental box. To avoid overheating in the box it was cooled by a small fan (Figure 5.5 c), which was driven by a 12-volt power supply.

In the first treatment groups, hatchlings were exposed to the unfiltered light. Then the dichroic filters were attached in front of the light source individually (Figure 5.7 a) to test aversion or preference of hatchlings for specific wavelengths transmitted. Additional testing was made by placing acrylic glass (5 mm) in front of the light source and filters to reduce overall illuminance (E_V) in the experimental box (Figure 5.7 b). The acrylic glass did not bias the transmission of light between 350 and 850 nm (WITHERINGTON and BJORNDAL, 1991a, NEUGEBAUER, pers. comm.). Illuminance in the two-choice box was measured separately for all filters at the starting point of the hatchlings, 85 cm from the source (Table 5.3). For this, a BEHA Unitest 9342 digital Luxmeter was used. This is the same device used in the field in the Belek nesting area (Chapter 4). Illuminance levels used in my indoor experiments were adapted to values which I measured in Belek in the field. Thus the illuminance levels tested on hatchlings represent values for nesting beaches that are seriously affected by light pollution. Spectral transmission of the light source and filters was measured with an Ocean Optics USB 2000 Spectrometer (Figures 5.8 a-f).
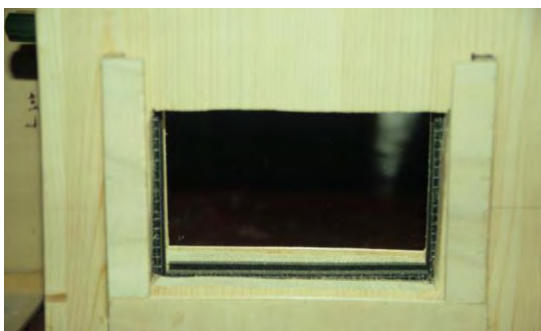

Figure 5.7 a: Dichroic filter attached in front of the Tungsten Halogen lamp

Figure 5.7 b: Acrylic glass placed in front of the Halogen lamp, for reducing illuminance in the box

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[23] ODL, SpA, Via Terzi di S. Agata 17, 24030 Brembate di Sopra, Italy

Table 5.3: Treatment groups tested for experimental lights

Light[1]	Low E_V treatment groups (lux)	High E_V treatment groups (lux)	Wavelengths (λ) transmitted
Tungsten Halogen lamp (10 watts) Unfiltered (white) light	Group 1 (7.89)	Group 2 (27.55)	350 – 850 nm (full spectrum)
Ultraviolet (UV)-transmitting filter (Filter code: Bronze 1)	Group 3 (2.06)	Group 4 (3.75)	< 400 and > 750 nm
Magenta filter (Filter code: SL4763)	Group 5 (4.42)	Group 6 (9.2)	< 450 and > 600 nm
Green filter (Filter code: WB5055)	Group 7 (3.31)	Group 8 (10.2)	500-550 and 700 nm
Yellow filter (Filter code LW520)	Group 9 (6.97)	Group 10 (21.5)	> 520 nm
Red filter (Filter code LW610)	Group 11 (4.48)	Group 12 (9.6)	> 610 nm

[1] In total 12 light-treatment groups were tested per species. The filter manufacturing codes are given by ODL, Italy. Illuminance (E_V) was measured with a BEHA Unitest 9342 digital Luxmeter. Wavelengths transmitted were measured with an Ocean Optics USB 2000 Spectrometer. The angle of the light beam passing the filters was 90 degrees for all measurements. Acrylic glass was used in the odd-numbered groups; no acrylic glass was used in the even-numbered groups.

Experimental lights and wave/surf sounds testing combined

Table 5.4: Treatment groups tested for experimental lights and sounds combined

Light (lux)[1]	Wave sound (dB SPL)	Surf sounds (dB SPL)
Unfiltered (white) (7.89)	Group 1 (44.0)	Group 2 (44.6)
UV-transmitting filter (2.06)	Group 3 (44.0)	Group 4 (44.6)
Magenta filter (4.42)	Group 5 (44.0)	Group 6 (44.6)
Green filter (3.31)	Group 7 (44.0)	Group 8 (44.6)
Yellow filter (6.97)	Group 9 (44.0)	Group 10 (44.6)
Red (4.48)	Group 11 (44.0)	Group 12 (44.6)

[1] In total 12 treatment groups per species were tested for lights and wave/surf sounds simultaneously. Additionally two groups were tested for 400 Hz at 40 db SPL (group 13) or 1000 Hz at 45.6 db SPL (group 14) and the unfiltered light at 7.89 lux simultaneously.
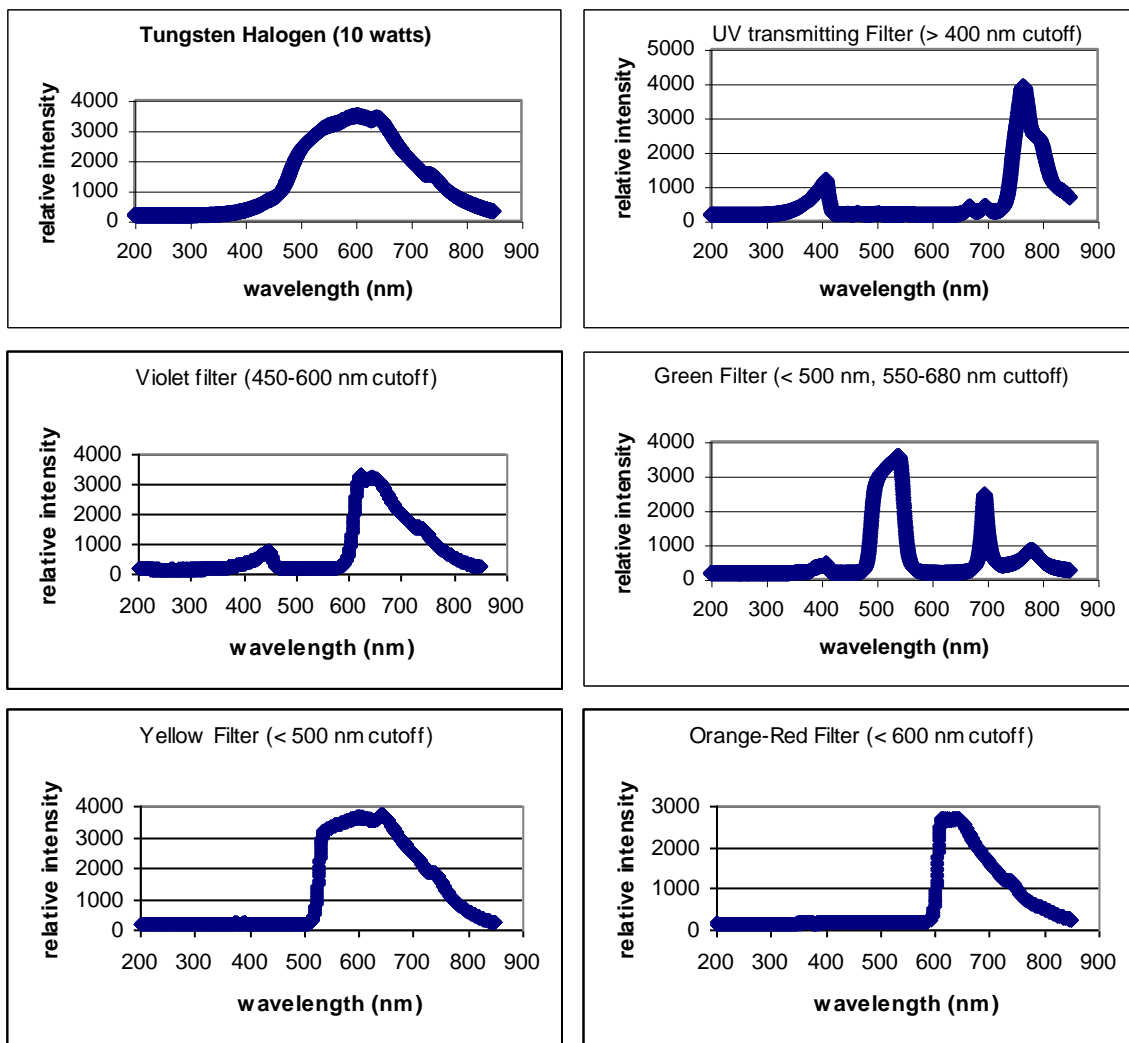
Figure 5.8 a-f (from upper left to lower right): Spectra emitted by the Tungsten Halogen lamp and transmitted by the filters. Measurements were made with an Ocean Optics USB 2000 Spectrometer. The angle of the light beam passing the filters was 90 degrees.

Statistics

Only hatchlings that were previously tested for increased activity were used for the experiments. Therefore hatchlings which did not move in the two-choice box within 5 minutes were assessed as indifferent to a specific stimulus (or stimuli). The percentage of indifferent hatchlings for each treatment group was calculated. Based on these data for single treatment groups, the Mean Indifference Rate (MIR) was calculated for the main test groups (Table 5.1) and for both species separately. The MIR is a term defined in this thesis according to:

Individual hatchling which did not move within 5 min in a treatment group ➔ hatchlings which did not move in one **treatment group** (%)➔ hatchlings which did not move in **main test groups** = MIR (%).

Thus, the MIR represents the overall ratio of indifferent hatchlings (not moved from the start position) in my indoor experiments.

For hatchlings showing crawling activity in the experimental box, statistical tests for binomial distribution (Sign test), Student's t-test, and FISHER'S Exact test were made.

Table 5.5: Statistical tests used for the evaluation of the indoor experiments (Setup 1)

Statistical test[1]	Assessment	H_0	H_1
Mean Indifference Rate (MIR)	Ratio of indifferent hatchlings (no movement) in the experimental box (after 5 minutes)	Not applicable (descriptive statistics)	Not applicable (descriptive statistics)
Sign test	Assessing binomial distribution of hatchlings in the experimental box for <u>one</u> treatment group	Hatchlings <u>equally</u> distributed in one treatment group	Hatchlings <u>unequally</u> distributed in one treatment group
Student's t-test for correlated samples	Comparing means of two samples in <u>one</u> treatment group	<u>No difference</u> in hatchling distribution in side 1 or side 2 of the two-choice box	<u>Difference</u> in hatchling distribution in side 1 or side 2 of the two-choice box
Fisher's Exact test (1)	Comparing distribution of hatchlings in the experimental box for <u>two</u> treatment groups	<u>No difference</u> in hatchling distribution comparing the same magnitude of a stimulus on <u>two species</u>, *C. caretta* and *C. mydas*	<u>Difference</u> in hatchling distribution comparing the same magnitude of a stimulus on <u>two species</u>, *C. caretta* and *C. mydas*
Fisher's Exact test (2)	Comparing distribution of hatchlings in the experimental box for <u>two</u> treatment groups	<u>No difference</u> in hatchling distribution comparing <u>two magnitudes</u> of the same stimulus (SPL or Illuminance) on the same species	<u>Difference</u> in hatchling distribution comparing <u>two magnitudes</u> of the same stimulus (SPL or Illuminance) on the same species

[1] All probabilities were assessed two-tailed. Significance level for H_1: $p < 0.05$, high significance level for H_1: $p < 0.005$. Statistics for binominal distribution (Sign test), Student's t-test and Fisher's Exact test were made using VassarStats statistical computation software.

5.1.2 Circular arena experiments (Setup 2)

Hatchlings previously tested in the two-choice box (Setup 1) were also tested in an outdoor circular arena for:

1) <u>Hatchling activity</u>. Hatchlings that showed indifference (no movement) in the two-choice box (Setup 1), <u>and</u> also did not move in the circular arena under outdoor conditions within 10 minutes, were assessed as physically inactive. This test was done to verify findings in particular for the sound treatment group, in which a large proportion of hatchlings did not move in the two-choice box (high MIR).

2) <u>Hatchling sea-finding ability</u>. It was tested if the treatments in the two-choice box (Setup 1) had an effect on later sea-finding ability of hatchlings. Comparison of hatchling crawling directions was made using statistics for circular distribution (Watson-Williams F-test).

Hatchlings were tested under outdoor conditions in a circular arena 4 m in diameter, which was drawn into the sand above the tide line, 10 m from the seashore. The selection of this setup is based on similar experiments conducted with hatchlings (SALMON, 2003, SALMON and WITHERINGTON, 1994, SALMON et al., 1995b). My circular arena was divided into eight identical sectors of 45 degrees by wooden marks. Hatchlings were placed in the centre of the circle in groups of 2-4. Collective testing in the circular arena did not bias orientation behaviour of hatchlings. This is supported by observations in comparable experiments (VERHEIJEN and WILDSCHUT, 1973, MROSOVSKY and SHETTLEWORTH, 1975). The seashore was clearly visible from the position of the circular arena. The heading direction of hatchlings within the circle was observed from a procumbent position behind the test arena. In total, 71 treatment groups of Setup 1 (36 *C. mydas* and 35 *C. caretta* groups) plus one control group per species were tested. For three treatment groups (one *C. mydas*, two *C. caretta*, see Appendix 4) the outdoor experiments could not be conducted due to adverse weather effects (heavy rainfall). These hatchlings were released to the sea without being tested in Setup 2.

Statistics

For hatchlings showing crawling in the circular arena, each individual's exit point in one of the eight sectors was noted. The outcomes are listed in Appendix 4. Statistical evaluation was made using Oriana 2.0 for circular statistics. For each treatment group the mean vector and the length of mean vector were calculated.

Annotations: The <u>mean vector (MV)</u> indicates the direction in which majority of hatchlings were crawling. This is the preferred heading direction of a treatment group. The more the MV differs from straight seawards (0^0) the more the hatchlings are oriented landwards. A MV deviation of $\pm 20°$ from this angle indicates hatchling disorientation (SALMON, 2003). The <u>length of mean vector (LMV)</u> indicates the degree of hatchling dispersion within the circle. The higher the LMV (maximum value = 1), the higher is the proportion of hatchlings moving in the same direction. Thus a low LMV indicates higher dispersion of hatchlings in the circle. An LMV < 0.9 indicates hatchling disorientation (SALMON, 2003).

In addition, all hatchlings of a treatment group that showed no activity within 10 minutes in the circular arena were counted. Based on these data for single treatment groups, the overall percentage of inactive hatchlings was calculated for the three main test groups (sounds, lights, sounds and lights tested combined).

Example:

In one sound treatment group (*C. caretta* tested for 400 Hz at 40.8 dB SPL), 14 hatchlings oriented seawards within 10 minutes. Of these active hatchlings, six left the exit points of the circular arena between 315^0 and 0^0 ("mean sector 337.5^0"), eight hatchlings between 0^0 and 45^0 ("mean sector 22.5^0").
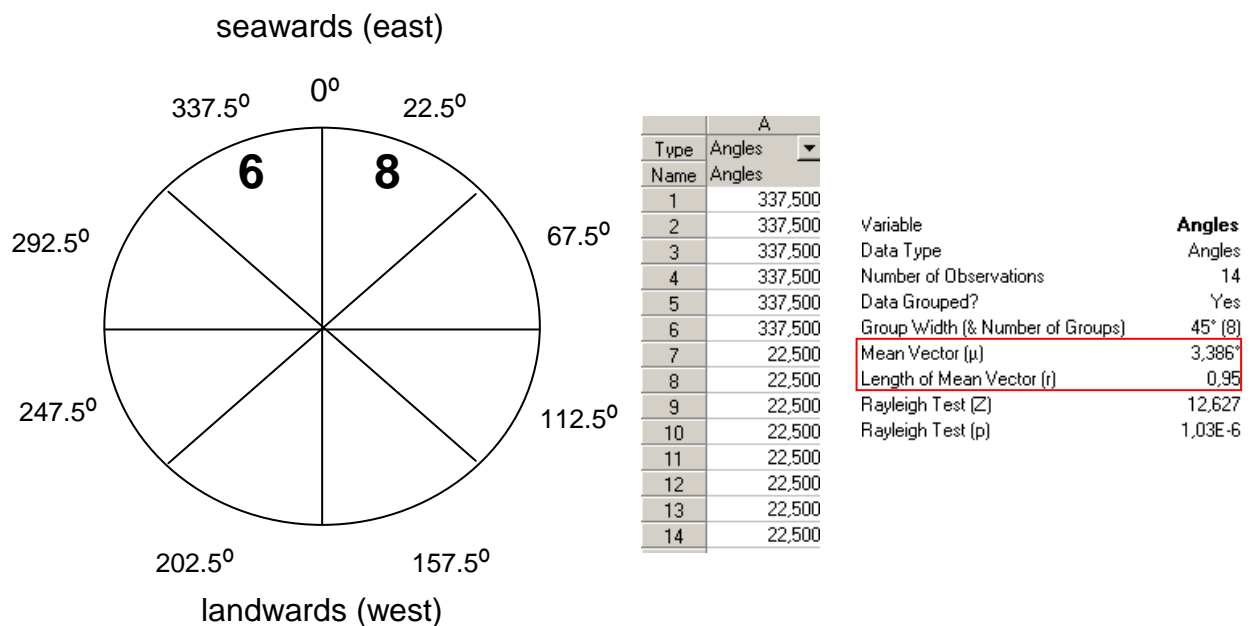

Figure 5.9: Hatchling orientation within the circular arena (example). Six hatchlings left the circle in sector 337.5^0; eight hatchlings left the circle in 22.5^0. Right: Data entry and calculation of mean vector (MV) and length of mean vector (LMV) for this treatment group in Oriana 2.0.

	A: *C. mydas*: sounds	B: *C. caretta*: sounds	C: *C. mydas*: lights	D: *C. caretta*: lights	E: *C. mydas*: sounds/lights	F: *C. caretta*: sounds/lights
	A	B	C	D	E	F
Type	Angles ▼	Angles ▼	Angles ▼	Angles ▼	Angles ▼	Angles ▼
Name	Angles	Angles	Angles	Angles	Angles	Angles
1	1,600	3,400	1,600	9,100	17,300	0,000
2	0,000	352,100	3,000	5,900	0,000	354,100
3	0,000	1,600	320,400	348,600	0,000	3,000
4	11,700	3,400	321,800	301,400	343,700	319,000
5	5,300	354,100	25,100	355,300	1,000	8,800
6	16,500	351,000	3,000	0,000	351,500	5,900
7	0,000	4,700	33,700	31,800	358,200	358,200
8	354,000	340,900	8,800	2,600	22,500	0,000
9	0,000	357,800	5,900	342,500	356,100	3,000
10	339,000	10,100	345,800	337,500	0,000	0,000
11	1,400	0,000	0,000	355,300	1,400	0,000
12			3,400	0,000	3,000	3,000
13					354,100	

Variable	Angles	Angles	Angles	Angles	Angles	Angles
Data Type	Angles	Angles	Angles	Angles	Angles	Angles
Number of Observations	11	11	12	12	13	12
Data Grouped?	No	No	No	No	No	No
Group Width (& Number of Groups)						
Mean Vector (µ)	0,89°	358,116°	359,621°	354,569°	0,642°	358,156°
Length of Mean Vector (r)	0,987	0,991	0,937	0,938	0,986	0,978

Figure 5.10: Data entry and calculation of second order mean vector (MV) and length of mean vector (LMV) for the main test groups (A-F) in Oriana 2.0. The first order MV, which was calculated for the single treatment group in the above example (Figure 5.9), is marked red.

The mean vectors (MV) calculated for <u>single treatment groups</u> were pooled and entered into a new table for calculating the second order MV for the respective <u>main groups</u> (three per species, Figure 5.10). The <u>Rayleigh test</u> (ZAR, 1999) was performed in Oriana 2.0 to determine if the main groups showed significant seaward orientation (0° in the circle) according to: H_0: hatchlings not oriented seawards, H_1: hatchlings oriented seawards (Significance level: $p < 0.05$).

Based on the data for main test groups, a <u>Watson-Williams F-test</u> (BATSCHELET, 1981) was performed in Oriana 2.0 to test if the mean vectors of the main test groups differed significantly from each other according to: H_0: hatchling orientations in main groups do not differ from each other, H_1: hatchling orientations in main groups differ from each other (Significance level: $p < 0.05$).

5.2 Results

5.2.1 Two-choice box experiments (Setup 1)

Hatchling movement patterns

As outlined in Chapter 5.1, hatchlings showed different behavioural patterns in the experimental two-choice box.

A) <u>Indifference</u>: hatchlings put in the box did not show any activity and remained motionless at their starting position, facing the vertex of the box. In some cases hatchlings showed rotation around their own axis but still did not crawl.

B) <u>Attraction for stimulus</u>: hatchlings turned and crawled towards the stimulus. First hatchlings straightened their heads, orienting. This phase took up to 4 minutes before moving. Then hatchlings turned away from the vertex at their starting point, rotating towards the direction of the attracting stimulus, followed by approaching the stimulus, while pausing at certain intervals. Overall crawling speed was assessed as higher for *C. mydas* compared with *C. caretta*. Most *C. mydas* hatchlings reached a stimulus set at 85 cm distance from the starting point within 1-3 minutes, whereas *C. caretta* needed slightly longer, due to pausing more often. Some hatchlings of both species tried to crawl further towards the stimulus, but where stopped either by the wire netting, which protected the speaker, or by the acrylic glass/filter attached in front of the light source. In other cases hatchlings stopped in front of the stimulus and remained motionless until they were removed from the box (Figures 5.11 a, b).

C) <u>Aversion to stimulus</u>: hatchlings turned away from the stimulus and crawled towards the opposite direction of the two-choice box, which was the darker side (light treatments), the more silent side (sound treatments), or the competing stimulus (sound/light treatments). The movement patterns are analogous to the ones described above.

When testing the Yellow filter and the Red filter in the light treatment groups, some hatchlings of both species showed circling; this is a shift in direction in the experimental box. These hatchlings initially crawled towards the light but turned around half way and moved to the opposite side of the box. Circling was not

observed for other light treatment groups (white light, UV, Magenta and Green filters used) or any sound treatment groups. Circling was also observed occasionally for *C. caretta* and *C. mydas* hatchlings which were tested for the unfiltered (white) light and wave/surf sounds simultaneously. These hatchlings first moved to wave/surf sounds but consequently turned to the light stimulus.

Figure 5.11 a: *C. caretta* hatchling crawling towards light source

Figure 5.11 b: *C. caretta* stopped in front of the illuminated window

Control group

The Mean Indifference Rate (MIR) was low for both species' control groups (no light/sound stimuli). Only 20% of the *C. mydas* and 12.5% of *C. caretta* hatchlings did not move away from their starting point within five minutes. Most hatchlings started crawling and showed equal distribution in the two-choice box (Sign test: H_1 with p = 0.77 for *C. mydas* and p = 1 for *C. caretta*).

Treatment groups tested for sounds

In this test group, hatchlings were tested according to Table 5.2. In contrast to the control groups, there was a considerable number of hatchlings in the sound treatment groups that did not move from their starting point. Consequently the MIR was high, with 71.5% calculated for 11 *C. caretta* treatment groups and 51% for 11 *C. mydas* treatment groups. Remaining hatchlings that moved showed equal distribution in the box (Sign test). Only two sound treatment groups were unequally distributed in the experimental box: *C. mydas* tested for 400 Hz sound at 40.8 dB and *C. caretta* tested for surf sound at 55.1 dB. In both cases hatchlings significantly oriented away from these sound stimuli (Figures 5.12 a, b). The Student's t-test confirmed that most *C. mydas* hatchlings were equally distributed in the experimental

box. These hatchlings did not significantly prefer the test sounds over the opposite (silent) side of the test box. In contrast, *C. caretta* hatchlings that moved, highly significantly preferred the side of the box that provided no sound stimulus. Comparing different species with the FISHER'S Exact Test (1) (see Table 5.5), it turned out that hatchling orientation differed significantly only for surf sound tested at 44.6 dB. Whereas *C. mydas* oriented significantly towards surf sound, *C. caretta* did not and was randomly distributed in the experimental box. Comparing different sound pressure levels (SPL) for one species with the FISHER'S Exact Test (2), no difference in distribution was found for any treatment group. The species were equally distributed in the two-choice box at both SPL tested.

Altogether, in the sound treatment groups most hatchlings did not show clear aversion to, or preference for the test sounds (Figures 5.6 a-f), and were randomly distributed in the two-choice box. The remarkably high MIR found in the sound treatment groups will be discussed in Chapter 5.3.
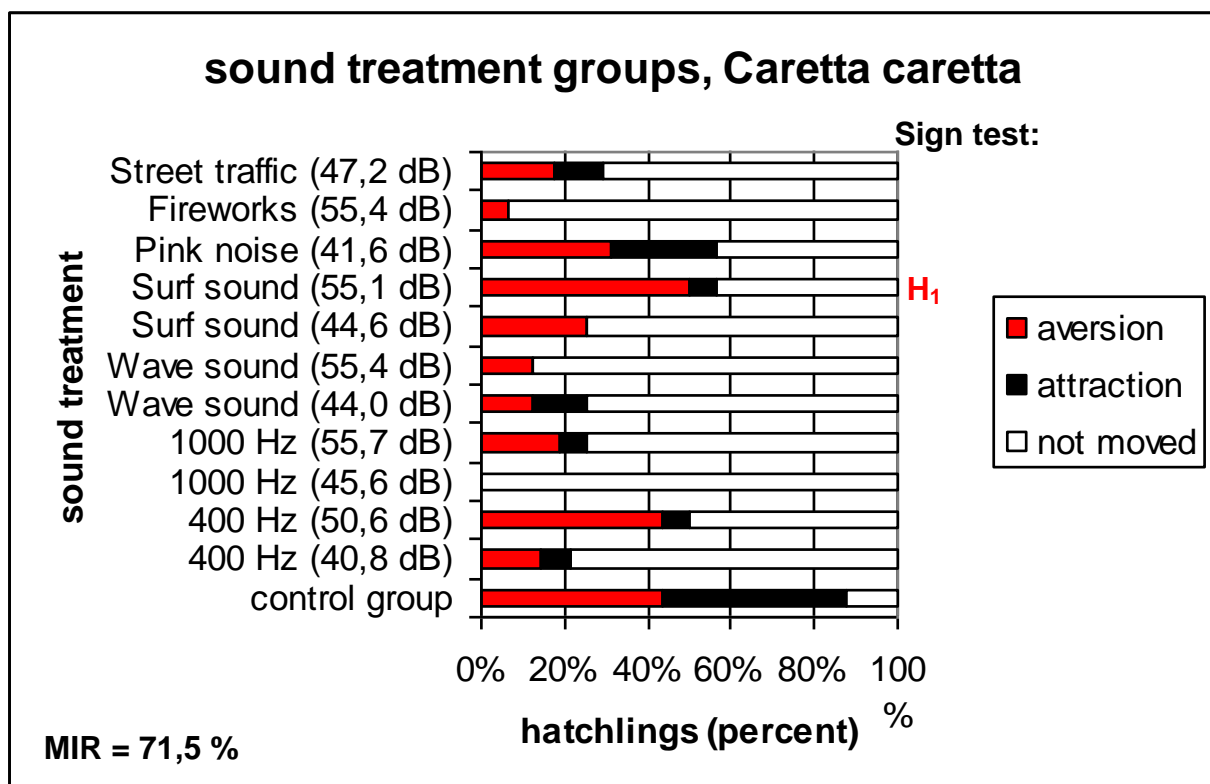

Figure 5.12 a: Results for sound-treatment groups, *C. caretta* (Setup 1). H_1 indicates significant aversion to surf sound at 55.1 dB SPL (Sign test). Note high Mean Indifference Rate (MIR) in all sound-treatment groups. These are hatchlings which did not move from their starting position. In contrast, the control group (no stimuli provided) showed equal distribution and higher hatchling activity compared with all sound-treatment groups.
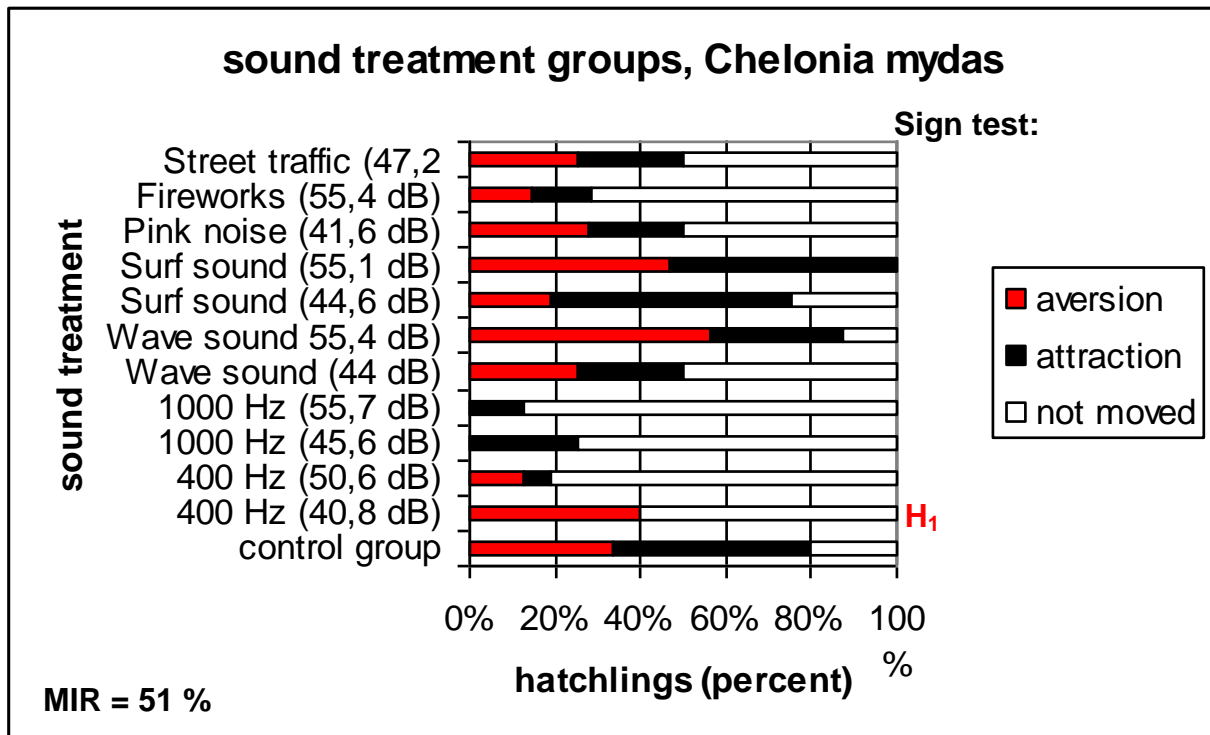
Figure 5.12 b: Results for sound-treatment groups, *C. mydas* (Setup 1). H_1 indicates significant aversion to the 400 Hz sound at 40.8 dB SPL (Sign test). Note high Mean Indifference Rate (MIR) in most sound-treatment groups. In contrast, the control group (no stimuli provided) showed higher hatchling activity compared with all but two sound-treatment groups (surf sound at 55.1 dB, and wave sound at 55.4 dB SPL).

Treatment groups tested for lights

In this test group (12 light-treatment groups per species), hatchlings were tested according to Table 5.3. The MIR calculated was 12.4% for *C. caretta* and 4.9% for *C. mydas*. This is far below the MIR calculated for the sound-treatment groups. In contrast to the sound-treatment groups, in most cases hatchlings started crawling and clearly showed orientation towards the provided (light) stimulus. The Sign test (see Table 5.5) revealed a significant (3 groups) or highly significant (19 groups) preference for the direction of light for all *C. mydas* and all but two *C. caretta* treatment groups. *C. caretta* hatchlings were only equally distributed in the experimental box when exposed to the Yellow filter, cutting off λ > 520 nm at 21.6 lux, and the Red filter, cutting off λ > 610 nm at 9.6 lux (Figure 5.13 a). The Student's t-test revealed all hatchlings' unequal distribution in the experimental box. Both species tested were highly significantly distributed in the direction of the lights. The FISHER'S Exact Test (1) revealed that *C. caretta* tested for the Yellow filter at 21.6 lux and the Red filter at 9.6 lux in fact showed differences in behaviour compared with *C. mydas*. Whereas *C. caretta* showed equal distribution in the box, interpreted as

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lesser attraction by the light source, *C. mydas* showed a preference for the Yellow and Red filters at high illuminance levels. This confirms the outcomes of the Sign test. For the other treatment groups no difference in distribution was found when comparing the two species. The FISHER'S Exact Test (2) revealed that *C. caretta* showed differences in behaviour when tested with the Yellow filter and the Red filter at different illuminance levels. Whereas this species showed a preference for the filtered light at lower illuminance (Yellow filter: 7 lux, Red filter: 4.5 lux), it showed equal distribution at higher illuminance (Yellow filter: 21.5 lux, Red filter: 9.6 lux). For the other treatment groups, no difference in distribution was found when comparing two illuminance levels per filter tested.

Altogether, hatchlings of both species were attracted by the white light source, which had broadband properties with a peak at 600 nm (Figure 5.8 a). All hatchlings highly significantly chose this light source over the dark side of the two-choice box. Only *C. caretta* showed less attraction to the full spectrum light at highest illuminance (27.6 lux). Both species' hatchlings also highly significantly chose the UV-transmitting filter over the dark side. This filter transmitted wavelengths from 350 to 400 nm and cut off the visible range between 400 and 720 nm (Figure 5.8 b). All hatchlings highly significantly oriented towards the Magenta filter. This filter transmitted between 400 and 450 nm and cut off wavelengths between 450 and 600 nm. This filter also showed distinct peaks in the higher spectrum between 600 and 800 nm (Figure 5.8 c). All hatchling orientation towards the Green filter was highly significant. This filter transmitted wavelengths from 500 to 550 nm and also 700 nm (Figure 5.8 d). The Yellow filter, cutting off wavelengths below 520 nm (Figure 5.8 e), showed differences in hatchling orientation. Whereas in the low-illuminance (7 lux) treatment group, *C. caretta* hatchlings were highly significantly attracted, *C. caretta* hatchlings were not significantly attracted in the high-illuminance (21.5 lux) treatment group, which is evidence for xanthophobia at higher brightness levels. In contrast, *C. mydas* was attracted to the Yellow filter in all cases, which is evidence for positive phototaxis. The Red filter (cut off < 610 nm, Figure 5.8 f) also showed differences in the orientation behaviour of the two species. Whereas the Red filter attracted *C. mydas* with high significance ($p < 0.005$) at both illuminance levels in the box (4.5 and 9.6 lux), *C. caretta* was only significantly ($p < 0.05$) attracted by this filter at lower illuminance and showed equal distribution in the higher illuminance treatment group.
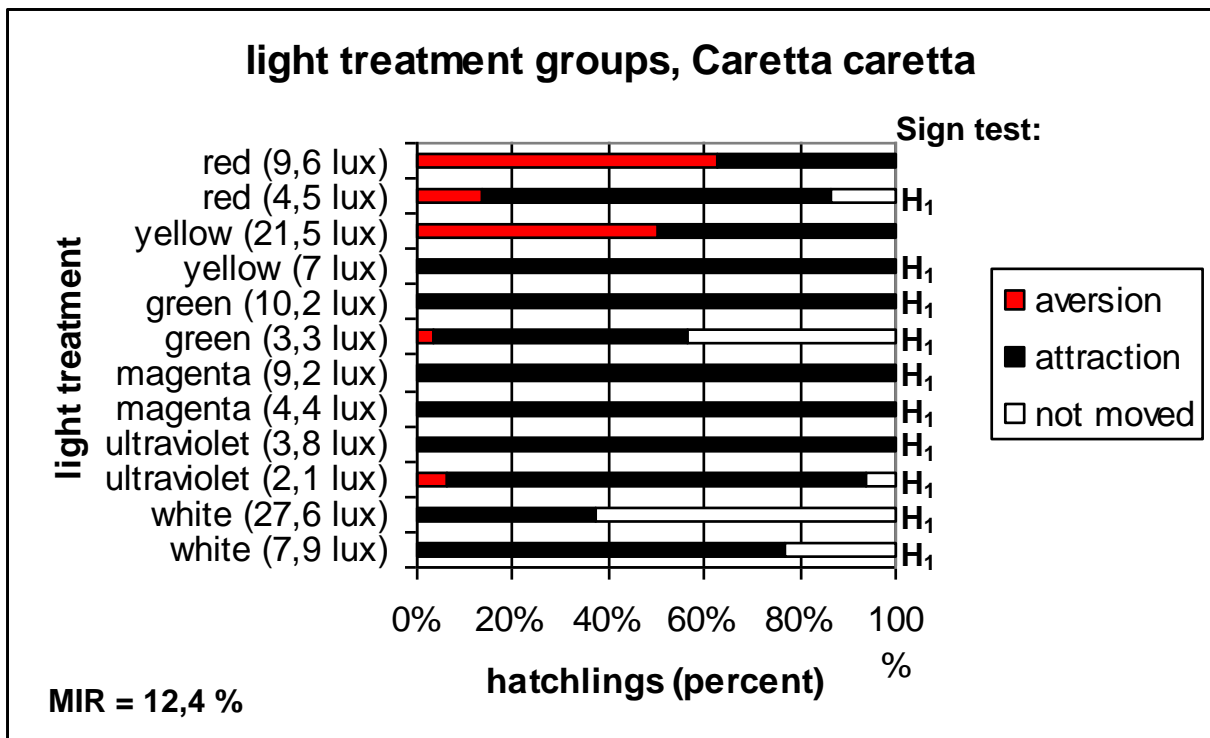
light treatment groups, Caretta caretta

Figure 5.13 a: Results for light-treatment groups, *C. caretta* (Setup 1). H₁ indicates significant attraction to the light source and most colour filters tested (Sign test). Increased aversion to the Red filter at 9.6 lux and the Yellow filter at 21.5 lux was observed in this species. Note low Mean Indifference Rate (MIR) = high hatchling activity, compared with the sound-treatment groups (Fig. 5.12 a).
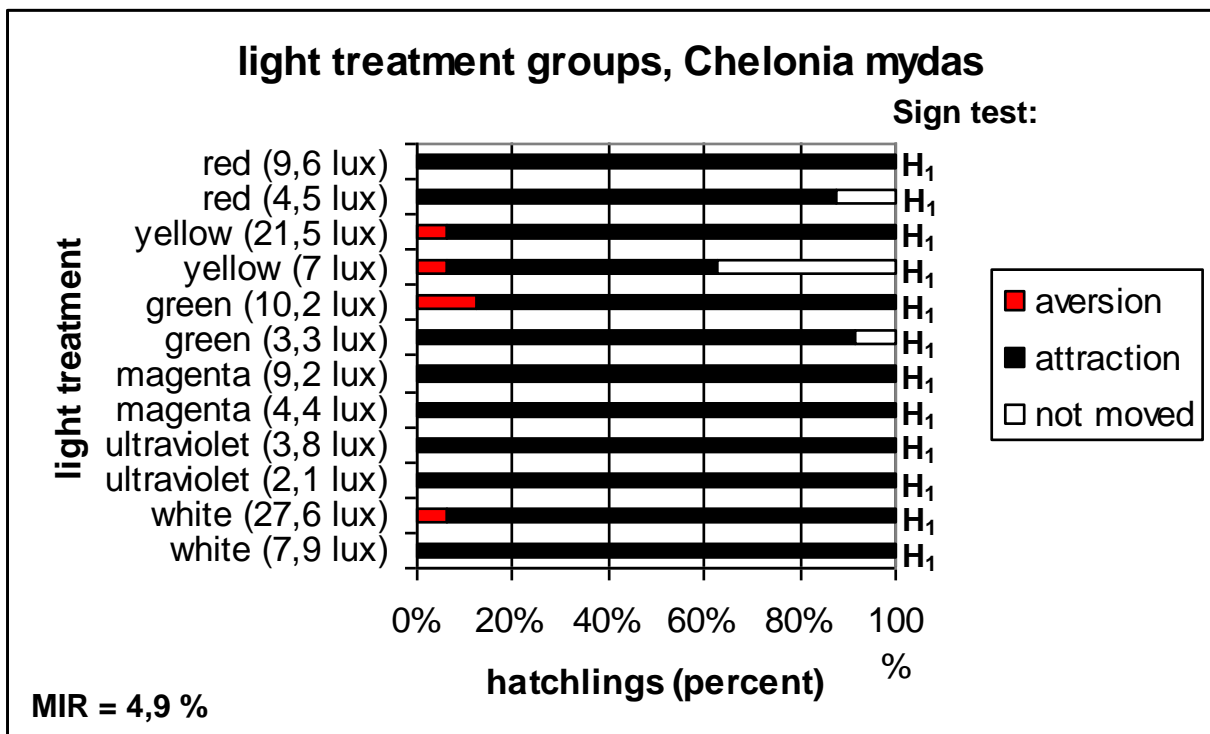

light treatment groups, Chelonia mydas

Figure 5.13 b: Results for light-treatment groups, *C. mydas* (Setup 1). H₁ indicates significant attraction to the light source and all colour filters tested (Sign test). Note low Mean Indifference Rate (MIR) compared with the sound-treatment groups (Fig. 5.12 b) and high attraction to the Yellow and Red colour filters, compared with *Caretta caretta* (Fig. 5.13 a).

Treatment groups tested for sounds and lights

In this test group, sound and light stimuli were tested simultaneously on hatchlings, according to Table 5.4. Twelve treatment groups per species were exposed to wave and surf sound recordings (Figures 5.6 c, d), emanating from the speaker, and the Tungsten lamp or one of the five dichroic filters attached (UV, Magenta, Green, Yellow, and Red). Other hatchlings were exposed to 400 Hz and 1000 Hz tones and the Tungsten lamp to test if hatchling behaviour differed from those treatment groups using wave and surf sounds. The MIR, measured for a total 14 treatment groups per species, was low, with 5.8 percent for *C. caretta* and 6.5 percent for *C. mydas*. This is comparable with results for the light-treatment group and far below the measurement for the sound-treatment groups. Testing hatchling distribution in two-choice box revealed that most treatment groups preferred the light to the sound source (H_1= true, Sign test). Both species always showed an attraction towards the white light, the UV-transmitting filter, the Magenta Filter, and the Green Filter. But only *C. mydas* showed an attraction towards the Red Filter. In total, six treatment groups (*C. caretta*: four groups, *C. mydas*: two groups) were equally distributed in the two-choice box. These were *C. caretta* tested for the Red and the Yellow filter and wave/surf sounds, and *C. mydas* tested for the Yellow filter and wave/surf sounds. These hatchlings showed unequal distribution in the experimental box (Student's t-test). The FISHER'S Exact Test (1) revealed a significant difference in both species' orientation in the presence of the Red filter and wave test sounds. Whereas *C. caretta* showed equal distribution in the box, *C. mydas* showed a preference for this light source. For all other treatment groups no difference in distribution was found when comparing the two species. Altogether, my findings confirm the outcomes of the previous test groups, testing lights and sounds exclusively. Whereas most lights had an attraction effect on hatchlings, wave/surf test sounds probably did not. This was also observed when testing artificial sounds (400 Hz and 1000 Hz tones). However, I found a difference compared with the outcomes of the test groups in which I used lights solely. In the light-treatment group, *C. caretta* showed an equal distribution in the presence of the Yellow filter, whereas *C. mydas* always showed attraction. But in the light/ sound-treatment groups, the overall number of both, *C. caretta* and *C. mydas* hatchlings, being equally distributed in the box, was higher (Figures 5.14 a, b). Thus, in this last setup I was able to prove a certain degree of aversion to the Yellow filter (xanthophobia) also in *C. mydas* (Figure 5.14 b).
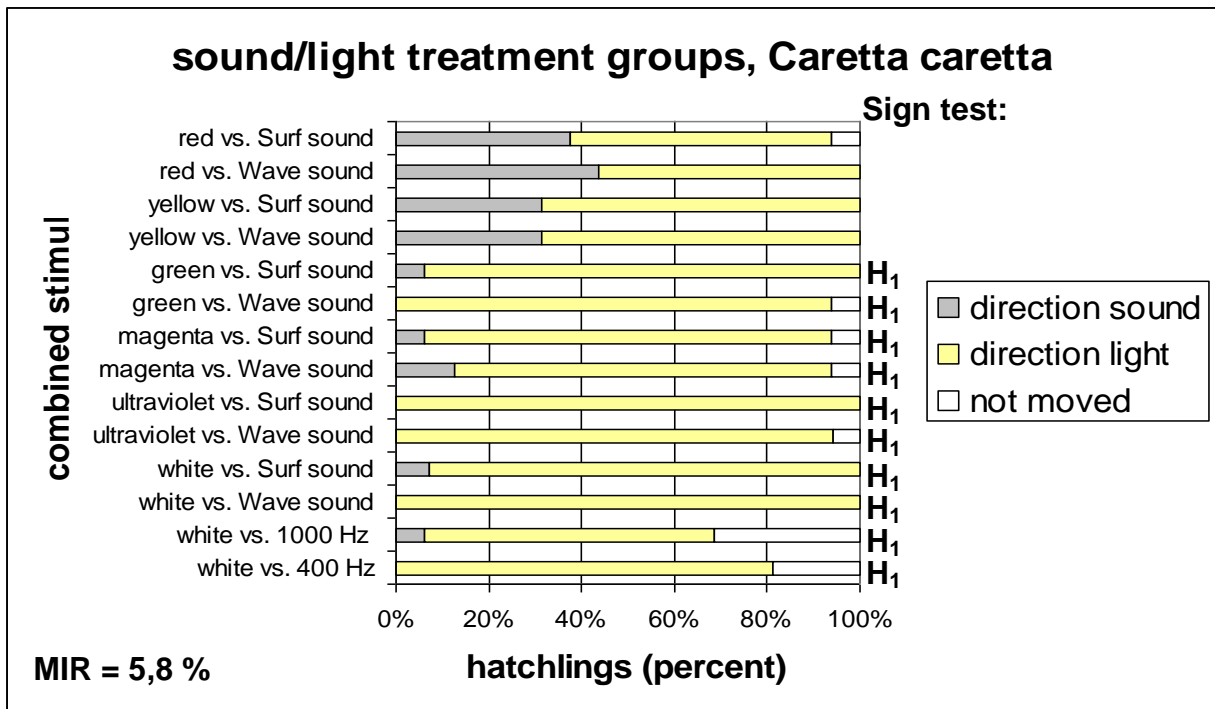
Figure 5.14 a: Results for sound/light-treatment groups, *C. caretta* (Setup 1). H_1 indicates significant attraction to the light source and most colour filters tested (Sign test). Increased aversion to the Red filter and the Yellow filter was observed in this species (moved away from light towards the sound stimulus provided). Note low Mean Indifference Rate (MIR) compared with the sound treatment groups (Fig. 5.12 a).
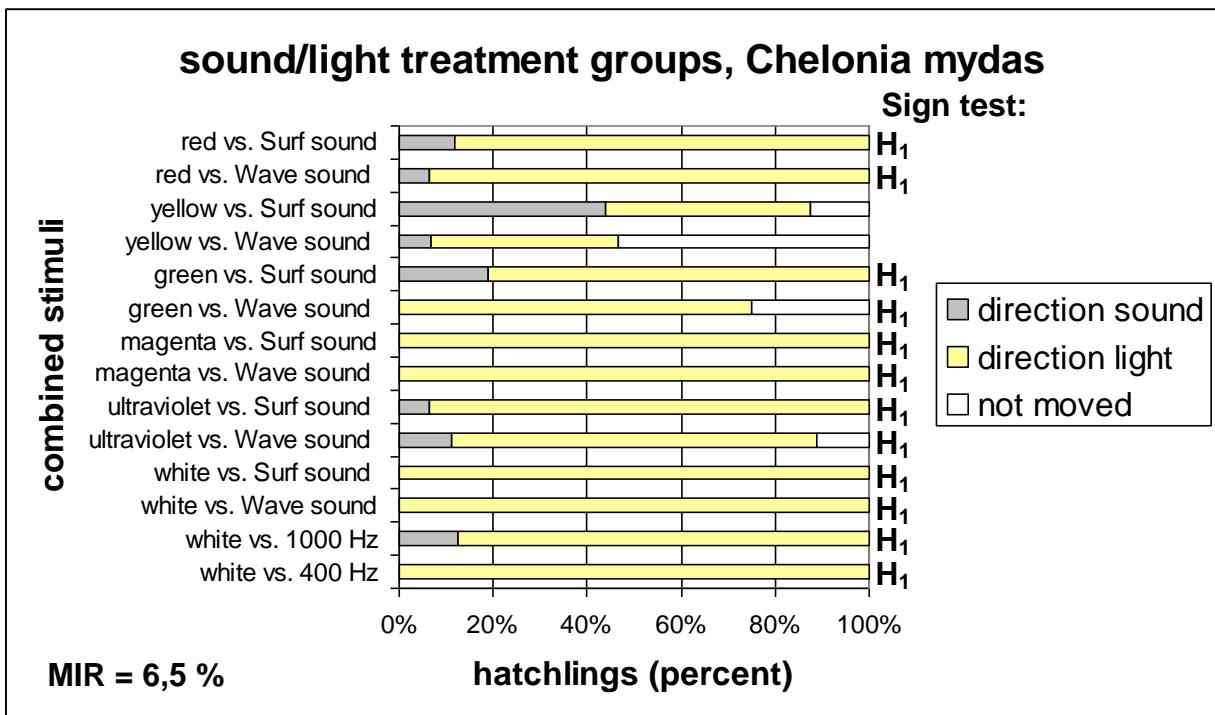

Figure 5.14 b: Results for sound/light-treatment groups, *C. mydas* (Setup 1). H_1 indicates significant attraction to the light source and most colour filters tested (Sign test). Increased aversion to the Yellow filter only was observed in this species (moved away from light towards the sound stimulus). Note low Mean Indifference Rate (MIR) compared to the sound treatment groups (Fig. 5.12 b).

5.2.2 Circular arena experiments (Setup 2)

Assessment of crawling activity

Nearly all hatchlings started crawling in the circular arena (Setup 2) within 10 minutes. *C. mydas* showed higher <u>crawling speed</u> in the arena, with up to 2 m/minute measured compared with *C. caretta* (1 m/minute). Moreover my findings revealed that <u>crawling activity</u> in the arena was slightly higher in *C. mydas* compared to *C. caretta*. These trends were consistent with my findings in the two-choice box (Setup 1). In both control groups all hatchlings started crawling (Setup 2). Overall crawling activity in the sound-treatment groups was 96% (*C. mydas* 99.5%, *C. caretta*: 93.5%); in the light-treatment groups 95.7% (*C. mydas*: 99%, *C. caretta*: 92.3%); and in the lights/sounds (combined) treatment groups 97.5% (*C. mydas*: 97.2 %, *C. caretta*: 96.8%). Thus crawling activity in the arena was consistently high within the control group and the three main groups. A comparison of hatchling inactivity in the outdoor setup with the Mean Indifference Rate (MIR) in the two-choice box reveals that there is a considerable inconsistency only for the sound-treatment groups. The MIR for both species was high in the sound-treatment groups, but the same hatchlings showed crawling on the beach. In contrast, crawling activity in both setups was constantly high in the light-treatment groups and the sound/light-treatment groups. This clearly shows that hatchling indifference in the sound-treatment groups was not based on inactive hatchlings, which will be discussed in Chapter 4.3.

Table 5.6: Comparison of Mean Indifference Rate (Setup1) and hatchling inactivity (Setup 2)

Main groups	Setup 1: *C. mydas* MIR (%) in the two-choice box	Setup 2: *C. mydas* inactivity (%) in the outdoor arena	Setup 1: *C. caretta* MIR (%) in the two-choice box	Setup 2: *C. caretta* inactivity (%) in the outdoor arena
Control group	20	0	12.5	0
Sound-treatment groups[1]	**51**	0.5	**71.5**	6.5
Light-treatment groups	4.9	1	12.4	7.7
Sound/light-treatment groups	6.5	2.8	5.8	3.2

[1] Note high MIR in the sound-treatment groups (Setup 1, marked red), which is contrary to the naturally high hatchling activity (frenzy) on the beach (Setup 2). High inactivity was not observed for any control or light-treatment groups in Setup 1.

Assessment of crawling directions

Control groups

Under bright ambient light conditions (nearly full moon), all hatchlings were significantly oriented seawards in the two control groups. A mean vector (MV) of 1.6° (*C. mydas*) and 351.2° (*C. caretta*) was measured. The length of the mean vector (LMV) was > 0.95 for both species (see Appendix 4). Thus, the MV and LMV calculated here are within the critical values (MV: less than $\pm$ 20° deviation from 0°, LMV: > 0.9) given for hatchling orientation on natural dark beaches (SALMON, 2003).

Treatment groups previously tested for sounds

In the sound group nearly all hatchlings oriented seawards at different ambient light conditions, ranging from no visible moon (cloudy sky) to nearly full moon. The highest deviation from straight seaward was observed for *C. mydas*, previously tested for wave sounds (55.4 dB SPL) in the two-choice box, with an MV of 339.3° and an LMV of 0.85 (Appendix 4). This treatment group showed high activity and was randomly distributed in the two-choice box (Setup1). Deviation from straight seawards was also observed for *C. caretta*, previously tested for a 400 Hz test sound (50.6 dB SPL). For this treatment group the MV was 340.9°, the LMV 0.88. This group also showed random distribution in the two-choice box. Comparison with the Rayleigh test revealed that all groups previously tested for sounds did not show abnormal behaviour on the beach and were significantly oriented seawards (Figures 5.15).
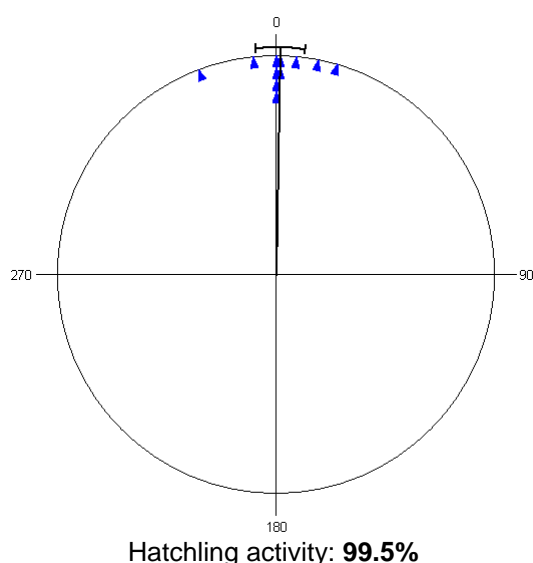

Hatchling activity: **99.5%**

Figure 5.15 a: Results for sound-treatment groups *C. mydas* (Setup 2)

Hatchling activity: **93.5%**
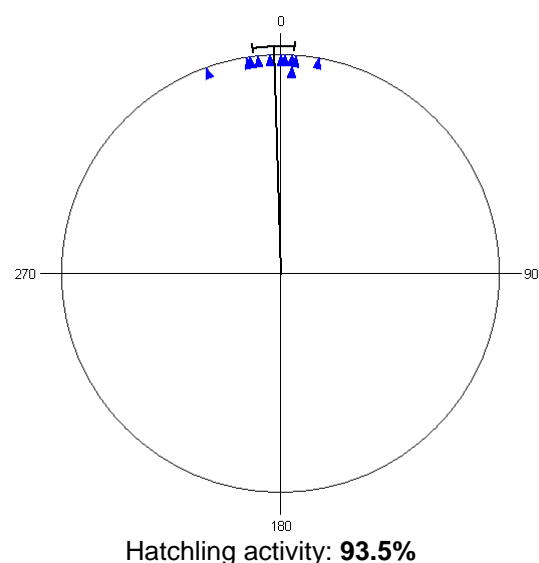
Figure 5.15 b: Results for sound- treatment. groups *C. caretta* (Setup 2)

Compared with the sound group, in the light group less hatchlings crawled straight seawards under comparable outdoor conditions. In total, nine treatment groups (five *C. mydas*, four *C. caretta*) showed an MV of more than ± 20° and/or an LMV of < 0.9 (Appendix 4). This indicates hatchling disorientation (SALMON, 2003). Remarkably high deviation from straight seaward was observed in treatment groups previously tested with the UV-transmitting filter at 2.1 and 3.75 lux. All *C. caretta* and *C. mydas* tested for this filter showed disturbed sea-finding in dark sky conditions (no moon visible) but some sky glow visible from the beach. Heading direction was consistent with the origin of the sky glow from a nearby town in the northwest (Chemuyil). Deviation from straight seawards was also observed for *C. mydas* previously tested for the Magenta filter (4.4 lux), the Green filter (3.3 lux), and the Red filter (9.6 lux), and for *C. caretta* previously tested for the Green filter (3.3 lux) and the Yellow filter (7 lux). When testing these groups no sky glow was visible on the beach, but dark cloudy sky at half moon and nearly full moon.

Comparison with the Rayleigh test revealed that in overall groups previously tested for lights did not show abnormal behaviour on the beach. All treatment groups were significantly oriented seawards, but with a higher angular dispersion compared with the sound-treatment groups (Figures 5.16).

Hatchling activity: **99%**
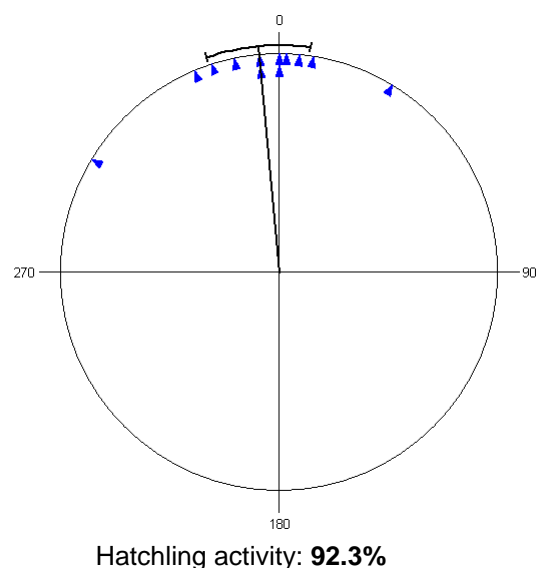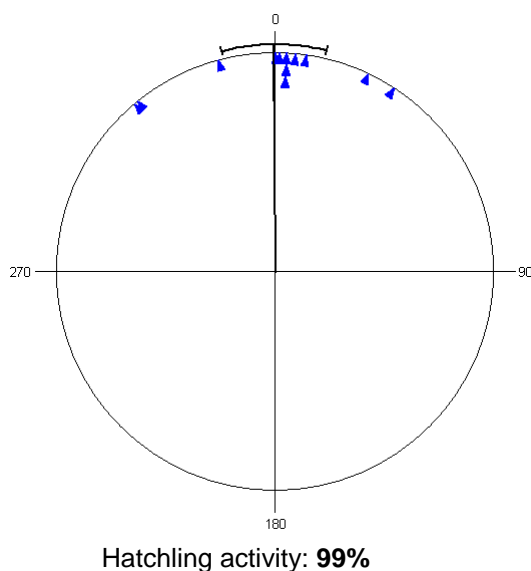

Hatchling activity: **92.3%**

Figure 5.16 a: Results light-treatment groups *C. mydas* (Setup 2)

Figure 5.16 b: Results light-treatment groups *C. caretta* (Setup 2)

Note higher angular dispersion for both species compared with the sound-treatment groups (Figs. 5.15).

<u>Treatment groups previously tested for sounds/lights</u>:

As for the light treatment groups, there was a higher proportion of hatchlings, which deviated from straight seawards in this group. In total five treatment groups (four *C. mydas*, one *C. caretta*) showed a MV of more than ± 20° and/or a LMV of < 0,9 (Appendix 4). This indicates hatchling disorientation (SALMON, 2003). High deviation from straight seawards was observed in the two *C. caretta* and *C. mydas* treatment groups that were previously tested for the UV-transmitting filter (2.1 lux) and surf sounds (44.6 dB SPL) simultaneously. In both cases the beach was completely dark due to a lunar eclipse (night of 27th Aug. 2007). Under dark beach conditions *C. mydas* also showed deviation from seawards when previously exposed to the Magenta filter (4.4 lux) and wave sounds (44 dB SPL). The same species showed deviation from straight seawards under nearly full moon conditions when previously tested for the Magenta filter (4.4 lux) and surf sounds (44.6 dB SPL), and the Green filter (3.3 lux) and surf sounds (44.6 dB SPL).

Comparison with the <u>Rayleigh test</u> revealed that in overall groups previously tested for sounds and lights simultaneously did not show abnormal behaviour on the beach. All treatment groups were significantly oriented seawards with a narrow angular dispersion (Figures 5.17).

Hatchling activity: **97.2%**
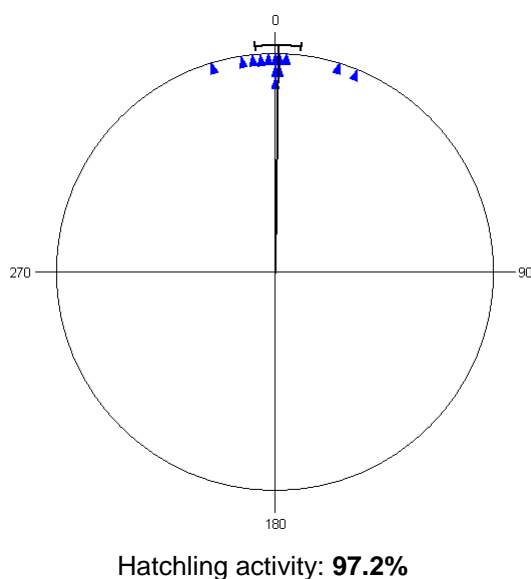
Figure 5.17 a: Results sound/light-treatment groups *C. mydas* (Setup 2)
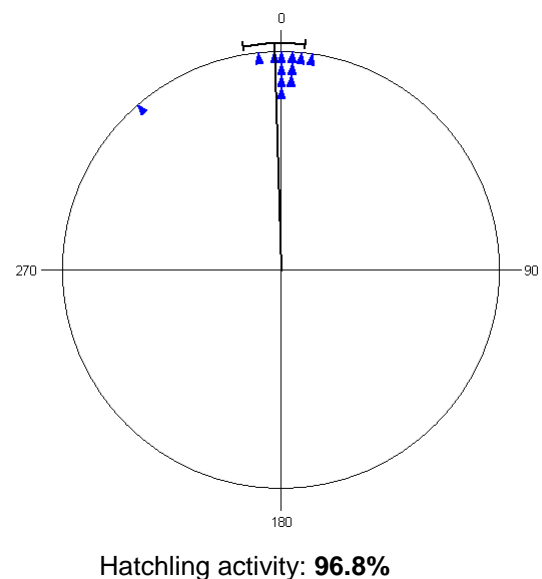

Hatchling activity: **96.8%**

Figure 5.17 b: Results sound/light-treatment groups *C. caretta* (Setup 2)

Note lower angular dispersion for both species compared with the light-treatment groups (Figs 5.16).

Table 5.7: Mean vector, length of mean vector, and Rayleigh test for Setup 2

Variable	A: *C. mydas*: sounds	B: *C. caretta*: sounds	C: *C. mydas*: lights	D: *C. caretta*: lights	E: *C. mydas*: sounds/lights	F: *C. caretta*: sounds/lights
	Angles	**Angles**	**Angles**	**Angles**	**Angles**	**Angles**
Data Type	Angles	Angles	Angles	Angles	Angles	Angles
Number of Observations	11	11	12	12	13	12
Data Grouped?	No	No	No	No	No	No
Group Width (& Number of Groups)						
Mean Vector (μ)	0,89°	358,116°	359,621°	354,569°	0,642°	358,156°
Length of Mean Vector (r)	0,987	0,991	0,937	0,938	0,986	0,978
Rayleigh Test (Z)	10,727	10,801	10,531	10,554	12,639	11,474
Rayleigh Test (p)	6,25E-7	8,35E-7	1,09E-7	1,51E-7	1,32E-6	1,46E-6

Table 5.7: for the main test groups sounds tested, lights tested, sounds/lights tested combined (one per species)

Comparing the main groups (A-F, see Tables 5.7, 5.8) per species with the Watson-Williams F-tests revealed that hatchling orientation did not differ between any of the groups. This indicates that the hatchlings in the main groups were significantly oriented in the same direction, which was orientation in a seaward direction.

Table 5.8: Watson-Williams F-tests for Setup 2

	A: *C. mydas*: sounds	B: *C. caretta*: sounds	C: *C. mydas*: lights	D: *C. caretta*: lights	E: *C. mydas*: sounds/lights	F: *C. caretta*: sounds/lights
A:	-----	0,47	0,858	0,375	0,951	0,567
B:	0,542	-----	0,829	0,608	0,509	0,993
C:	0,033	0,048	-----	0,569	0,878	0,84
D:	0,822	0,27	0,334	-----	0,364	0,62
E:	0,004	0,452	0,024	0,857	-----	0,589
F:	0,338	8,36E-5	0,042	0,252	0,3	-----

Table 5.8: Paired tests for main test groups (probabilities in upper half, F scores in lower half of the cross table)

5.3 Discussion

Most *C. caretta* and *C. mydas* hatchlings in the control groups (no stimuli) showed crawling and random distribution in the two-choice box (Setup 1) within five minutes. Preference for one of the two sides of the box was not observed here. Therefore a possible bias in the statistics was excluded here (WITHERINGTON and BJORNDAL, 1991a). The results show that the construction and installation of my two-choice box was appropriate, supporting the statistical power of my experiments. In the circular arena (Setup 2) all hatchlings of the control groups oriented straight seawards. This demonstrates that the nesting site Xcacel, Mexico, still had low light pollution levels in 2007, which justified my site choice for the behavioural experiments conducted here. In contrast, considerable light pollution was observed in Belek, Turkey, which probably would have biased the outcomes of these outdoor experiments. This is also supported by my previous experiments in the presence of artificial lights, in which a large proportion of *C. caretta* hatchlings oriented landwards (see Chapter 4).

Sounds:

When conducting my behavioural experiments using sounds, I faced the problem of adjusting the sound levels in the experimental box. This is because a test animal may simply not perceive a sound level set too low. Sound levels were adjusted according to a reference value of ± 50 db SPL, which is suggested to be the minimum aerial sound required to be perceived by *C. mydas* sea turtles (RIDGWAY et al., 1969, LENHARDT, 1994, see Chapter 1). For *C. caretta* there is no comparable electrophysiological data available using aerial sounds. Therefore I used the same sound levels as for *C. mydas* for testing also *C. caretta*. This was based on my assumption that *C. caretta* has a similar auditory sensitivity in the low-frequency range (< 1000 Hz) as *C. mydas*. Low frequency sound perception in *C. caretta* is confirmed by electrophysiological studies, but testing vibratory stimuli (BARTOL et al., 1999, BARTOL and Ketten, 2006). Moreover, I expected that both species' early life stages, the hatchlings, would not have a sensitivity to aerial sounds less than that of the *C. mydas* juveniles tested by RIDGWAY et al. (1969). Hatchlings' high sensitivity to aerial sounds is supported by preliminary data on hatchling ear anatomy (LENHARDT, 2005). As I aimed to imitate sound effects on hatchlings in the field, the sound levels used in experiment were also adjusted according to effective values, which I measured in the egg-laying zone of Belek (Chapter 4). When providing sound stimuli only (no lights), many hatchlings remained at their starting position. In total, half of

the *C. mydas* hatchlings (MIR: 51%) and 71.5 % of *C. caretta* tested for sounds did not move within 5 minutes. This is far above the Mean Indifference Rates calculated for the control groups and for both species that were tested in the presence of light stimuli. The high hatchling inactivity observed here is also inconsistent with my findings in Belek. In the presence of aerial sounds, but shielded against light, nearly all *C. caretta* hatchlings started crawling and were equally distributed in the experimental arena (see Chapter 4). So what is the reason for this high hatchling inactivity in the indoor Setup 1? One argument is that sound stimuli alone may not elicit crawling behaviour in hatchlings because they do not provide a definite orientation cue. In early studies it was demonstrated that sea turtles rely on visual cues for orientation. Blindfolded hatchlings were not able to find the sea under outdoor conditions (DANIEL and SMITH, 1947, CARR and OGREN, 1960, VAN RHIJN, 1979). But in my control groups all hatchlings started crawling in total darkness. This agrees with the hatchling behaviour observed in the *C. caretta* and *C. mydas* control groups (no stimuli) tested by WITHERINGTON and BJORNDAL (1991a). Preliminary findings of MANGIAMELE and LOHMANN (2005) also showed that *C. caretta* hatchlings started crawling in the presence of sound stimuli but in total darkness. In fact increased crawling behaviour is part of natural emergence patterns in hatchlings (WITHERINGTON et al., 1990). In this state of "frenzy", hatchlings show increased activity (WYNEKEN and SALMON, 1992). A decrease in ambient temperatures during the night (GLEN et al., 2005, WITHERINGTON et al., 1990), air drought, when taking hatchlings out of the covered buckets, as well as physical contact, elicited crawling behaviour in my test animals (pers. observation). Hatchlings' flipper movements, when testing for activity, evidenced this. Moreover, the two-choice box provided enough space and natural soil conditions (beach sand), probably also stimulating hatchlings. Thus, based on activity control, high sample sizes, and multiple measurements, it can be excluded that the high MIR in the sound-treatment groups is based on the use of inactive or weak hatchlings. Environmental stress, such as a rise of temperature to > 33° C in the nest, is known to cause the inhibition of movements in hatchlings and reduce hatching and emergence success (CHEEKS, 1997, FORTUNA and HILLIS, 1998, BLAIR, 2001). This thermal inhibition prevents hatchlings emerging during daytime, when sand temperatures are high (WITHERINGTON et al., 1990, MORAN et al., 1999), which would lead to desiccation with lethal outcomes. However, I kept hatchlings in a cool place and also conducted my experiments during the night,

when temperatures were lower. Therefore it can definitely be excluded that the temperatures in my two-choice box were over the threshold level for hatchling activity and caused inhibition. Overheating in the box was also prevented in the light-treatment groups by a cooling system (small ventilator) attached to the lamp. Hatchlings are also known to show reduced activity as a result of emergence stress. Studies have shown that hatchlings partly switch to anaerobic metabolism when digging their way out from underground nests, which might require resting near or at the surface to return increased lactate levels to normal (BALDWIN, 1989). High lactate levels are known to decrease behavioural capabilities and cause lethargy in reptiles, which would explain an inhibition of movement (BENNETT, 1982). However, increased lactate levels in my hatchlings should be excluded, as they were not exposed to emergence stress before my experiments but kept in a resting state in buckets permeable to air. Thus there must be other factors causing the inactivity of hatchlings in the sound-treatment groups.

It is suggested here that the high MIR in my experiments may be a consequence of disruption of natural crawling behaviour by increased sound levels in the experimental box, probably causing an inhibition of motor activity. Submerged freshwater turtles (*Chrysemys picta*) are known to avoid anthropogenic sounds (VOGT, 1980). Comparable data indicate that the ambient noise in coastal habitats disturbs submerged sea turtles (LENHARDT et al., 1996, SAMUEL et al., 2005). One of the possible effects observed in sea turtles is the loss of motor control (LENHARDT, 1994; LUTCAVAGE et al., 1997). Underwater, airguns fired at high sound pressure levels of 120 dB re 1 µPa at 1 m (O'HARA and WILCOX, 1990) and 175-179 dB re 1 µPa at 1 m (MOEIN et al., 1995) affect *C. caretta* sea turtles by eliciting avoidance reactions. MCCAULEY et al. (2000) reported erratic behaviour (abnormal swimming patterns) in caged *C. caretta* and *C. mydas* in the presence of airgun sounds at 166-175 dB re 1 µPa at 1 m. MOEIN et al. (1995) also measured increased stress levels in juvenile *C. caretta* and a temporary shift in hearing capabilities, which returned to normal only two weeks after testing. This latter study demonstrated that sound exposure had a lasting effect on the sea turtles. A temporary threshold shift (TTS) underwater could potentially prevent an individual from detecting predators or prey or man-made dangers, like boating traffic (SAUNDERS et al., 1985, LENHARDT, 1986). But do sounds also have an effect on sea turtles while on land?

My study investigated low-frequency sounds in the medium of air at sound levels of 40-55 dB re 20 µPa at 1 m. Previously I demonstrated that these aerial sounds are within the hearing range of sea turtles (Chapter 4). Air-conducted sound perception is also supported by recent findings on the inner ear of hatchlings (LENHARDT, 2005). In none of my sound-treatment groups did hatchlings show significant attraction towards sounds. In contrast, two treatment groups showed aversion, to 400 Hz at 40.8 dB (*C. mydas*) and to surf sound at 55.1 dB (*C. caretta*). These hatchlings significantly oriented away from these sound stimuli. For the artificial tone, this may be plausible, as sea turtles show high sensitivity to tones and vibrations at 400 Hz (RIDGWAY et al., 1969, BARTOL et al., 1999). Hence, hatchling behaviour may be interpreted as avoidance reaction. On the other hand, aversion to the broadband surf sound tested is unexpected, as these sounds are the basic component in sea turtles' coastal habitats and naturally should not have a disturbing effect. Altogether hatchlings neither chose the surf/wave sound stimuli over the opposite (silent and dark) side of the two-choice box, nor did they significantly orient away. Thus my results tend to be in agreement with the preliminary findings of MANGIAMELE and LOHMANN (2005), which is the only comparable study available at present. In total darkness, these authors' *C. caretta* hatchlings did not significantly orient towards aerial wave sounds. Besides aerial sounds, substrate vibrations were suspected as playing a possible role in hatchlings' seaward orientation in my thesis (Chapter 4). MANGIAMELE and LOHMANN (2005) did not specify if their setup also included vibrations. But my setup did; vibrations were propagated over the wooden construction of the two-choice box. Based on my results, I conclude that neither the aerial surf/wave sounds tested, nor the resulting vibrations in the box, elicits orientation behaviour (attraction) in *C. mydas* and *C. caretta* hatchlings. But vibrations must be considered as one possible cause of hatchling inhibition in my sound-treatment groups.

As for surf/wave sound, a large proportion of hatchlings of both species showed inactivity in the presence of artificial sounds (pure tones, fireworks, traffic sounds, and pink noise). This may be interpreted as motor arrest as a defence mechanism against sensory stimuli. Electrical stimulation of the brain (ESB) is known to immobilize mammalian test animals (DELGADO, 1969). Though not proven for sea turtles, stimulation by sounds or vibrations may have a comparable effect. I do not have electrophysiological data but behavioural evidence for this. A large proportion of hatchlings did not crawl in the sound-treatment groups (Setup 1) but the same

hatchlings showed crawling on the beach later (Setup 2). This is one argument that my test animals were in a normal active state, and supports the negative effect (motor inhibition) of test sounds on these otherwise active hatchlings. Based on these results, I propose a possible negative effect of sounds on sea turtle hatchlings on land. However, my results must be analysed carefully. In water, sea turtles displayed agitated behaviour, abrupt body movements, startle responses, and even inactivity at the bottom of a tank in response to low frequency stimuli (LENHARDT et al., 1983). In another study, LENHARDT (1994) applied infrasound signals to *C. caretta* in an underwater tank. He observed that turtles stayed near the water/air boundary, which may be interpreted as avoidance reaction. In this context it must be discussed whether the closed construction of my two-choice box enabled the test animals to clearly locate the origin of the test sounds. It is suggested that sound reflection within the box was high. Vibrations were propagated over the entire box (pers. observation). Moreover, I applied the sounds from close distance (85 cm). This is relevant, as a turtle detecting a disturbing sound from a greater distance may escape the noise (VOGT, 1980, O'HARA and WILCOX, 1990, MOEIN et al., 1995), whereas a caged turtle may not be able to retreat and show different behavioural patterns (MCCAULEY et al., 2000). Thus, it cannot be excluded that hatchlings in my closed setup, exposed to noise from close proximity, were not able to clearly locate its source and to retreat. This may explain my hatchlings' indifference, rather than avoidance reactions, e.g. movement to the more "silent" side of the box.

A lasting effect of sounds on hatchlings' behaviour, such as inhibition of movement or disorientation, could not be proven in my outdoor experiments (Setup 2). Most hatchlings in the sound-treatment groups significantly oriented straight seawards in the circular arena later, as did the control groups. Because the sound stimuli in Setup 1 were provided from a small distance, an auditory temporary threshold shift (TTS) may be possible in my test animals (MOEIN et al., 1995). However it is suggested here that a TTS probably would not have observable effects on hatchling orientation. Reduced sensitivity to sounds on land, caused by a TTS, may not affect hatchlings or nesting females, as both life stages rely primarily on their visual sense, which enables predator detection and also seaward orientation. Regarding developed nesting beaches, inhibition of hatchlings caused by high intensity sounds (noise pollution) is also questionable. First, the beach sand partly absorbs low frequency vibrations at greater nest distance to the shore. Second, noise pollution is correlated

with light pollution in developed areas (see Chapter 4). Thus, even if noise pollution had a negative effect by inhibiting movement, artificial lights from landwards would be even worse, counteracting this and attracting hatchlings. This was demonstrated in my experimental setup using sounds and lights combined (Figs 5.14 a, b). Sounds did not have an inhibitory effect on hatchlings in the presence of attracting light, the latter providing the stronger stimuli.

Nevertheless, my findings reveal that the effect of noise on sea turtles on land cannot be generally excluded. Though sound perception in adult sea turtles is decreased compared with in water and the thick tympanum of the middle ear also blocks air-conducted sounds (RIDGWAY, 1969, LENHARDT et al., 1985), preliminary findings of the National Aviation Service (1990) showed that *C. caretta* females nesting in Zakynthos, Greece, were significantly deterred by aircraft approaching the nearby airport. Hence, disturbance of females may be a major problem at nesting beaches, which makes the regulation of nighttime noise necessary for conservation purposes.

Lights:

In the light-treatment groups most hatchlings moved from their starting positions and started crawling. This was observed both in the indoor Setup 1, resulting in a low MIR, and the outdoor Setup 2. The high hatchling activity observed is consistent with the findings of WITHERINGTON and BJORNDAL (1991a), who tested *C. caretta* and *C. mydas* hatchlings in a two-choice box for adjustable light sources, using different narrow-band filters. Less than 10% of both species' hatchlings did not choose either side of the test box after two minutes in their setup. Comparable data confirm high hatchling activity under outdoor conditions. Less than 5% of *C. caretta* hatchlings did not move in a circular arena within five minutes (WITHERINGTON and BJORNDAL, 1991b) or two minutes (SALMON et al. 1995b). In contrast to hatchlings' indifference or random distribution in the sound- treatment groups, hatchlings were highly significantly oriented towards lights in most cases (Setup 1). This confirms the attracting effect which light of specific wavelengths and intensity has on hatchlings of both species (MROSOVSKY and CARR, 1967, MROSOVSKY and SHETTLEWORTH, 1968, SALMON and WITHERINGTON, 1994, WITHERINGTON and MARTIN, 1996, SALMON, 2003). In my experiments, both species were strongly attracted by the polychromatic (white) light source at high illuminance levels in the box (7.9 lux and 27.6 lux). These findings are consistent with the high hatchling disorientation observed close to Metal Halide lights, with their high illuminance levels (up to 40 lux), on the beach of Belek

(Chapter 4). The negative effect of polychromatic sources on females and hatchlings is confirmed in the literature (WITHERINGTON, 1992a, WITHERINGTON and MARTIN, 1996). Only one *C. caretta* treatment group showed less attraction to this white light source (27.6 lux) in Setup 1. This deviation was caused by lower crawling activity in this group, not the light source itself. This was verified when testing these hatchlings on the beach. More than 50% of these test animals did not move here, which was exceptional in my experiments (Setup 2) and may be caused by diseased or otherwise weakened hatchlings. With regard to hatchling activity, it was observed that *C. caretta* generally paused more often when orienting, whereas *C. mydas* were more agile. This interspecific difference was also described by WITHERINGTON and BJORNDAL (1991a). It is suggested here that increased crawling speed may be an adaptation to compensate for large nest distances from the shoreline. Whereas *C. caretta* nest close to the shore within 15-20 m, *C. mydas* nests are found up to 30 m landwards (ARENAS, pers. comm., pers. observation). Faster crawling may help *C. mydas* to avoid the increased risk from predators on land.

Both species also showed high attraction to the UV-transmitting filter at comparatively low illuminance levels in the box (2.1 lux and 3.8 lux). This filter cut off the visible range λ > 400 nm. The hatchling behaviour observed is consistent with literature data, suggesting the ability of sea turtles to perceive short visible wavelengths and also UV light. In behavioural experiment, WITHERINGTON and BJORNDAL (1991a) found that both *C. mydas* and *C. caretta* oriented towards near-ultraviolet light at 360 nm. Spectral transmission measurements of the ocular media of *C. mydas* showed that wavelengths at 325 nm were transmitted in the sea turtle eye (MÄTHGER et al., 2007). These authors concluded that this species is able to perceive light also in this lower UV spectrum, which is not visible to humans at all. LIEBMAN and GRANDA (1971) suggested that the ability to perceive UV light is an adaptation to sea turtles' underwater habitat, where they spend most time of their life. In this habitat it is advantageous to have a high sensitivity to the shorter visible spectrum of light, as these wavelengths are less absorbed than higher wavelengths. It is suggested here that UV perception may also serve sea turtles on land when orienting seawards. As UV rays have differing reflection from soil (10% reflected), dry beach sand (15%), and sea foam (25%), sea turtles may be able to quantify the proportion of UV being reflected by the ocean and use it as an orientation cue. But this is speculative for *C. caretta* and *C. mydas* hatchlings, as both species emerge

mainly at night (WITHERINGTON et al., 1990, GLEN et al. 2005) when UV levels are normally reduced. However, hatchling emergences in the early afternoon (CHAVEZ et al., 1968) and the late afternoon (WITZELL and BANNER, 1980) are reported in the literature. The perception of UV light may also enable females to return to the sea when nesting in daytime. Daytime nesting is not common for most species but confirmed for the Kemp's Ridley (*L. olivacea*) and the Olive Ridley (*L. olivacea*), both known for mass nesting events, called "arribadas" (MARQUEZ, 1990). Thus for daytime nesters, UV-perception ability may be a compensation for the lack of definite brightness cues and contrasts, which would normally guide sea turtles and hatchlings to the sea at night. Further investigation on the UV-perception capabilities of sea turtles is needed here.

Another phenomenon of turtle vision is the perception of polarized light. Polarized light has been suggested as a possible cue for water-finding in freshwater turtles (GIBBONS et al., 1983). The use of polarized light as an orientation cue is also possible for sea turtles. As light is partly polarized due to reflection from the water surface, hatchlings and females could detect it when orienting seawards. After entering the sea, it may also serve as a cue when navigating offshore. This is because polarized light is always perpendicular to the direction of the sun. Therefore sea turtles, swimming at the surface, may detect the position of the sun even during overcast conditions. However the role of light for offshore migration is speculative. Sea turtles are known to use the Earth's magnetic field for long distance migration (LOHMANN and LOHMANN, 1994), with their magnetic compass probably being light-independent (LOHMANN and LOHMANN, 1993).

In my experiments, hatchlings of both species oriented highly significantly towards the Magenta filter (9.2 lux and 4.4 lux). This is consistent with previous findings that hatchlings show high attraction to violet light at 400 nm (WITHERINGTON, 1997, WITHERINGTON and BJORNDAL, 1991a). This dichroic filter transmitted light with peaks in the 400 to 450 nm (violet), but also in the 600 to 750 nm (yellow-red) spectrum, therefore it is ambiguous which portion elicited attraction in hatchlings. It is suggested here that at least *C. caretta* hatchlings must have responded to the peaks in the violet range, as they significantly showed less attraction for the yellow-red portion in subsequent experiments.

Both species were also significantly oriented to the Green filter (3.3 lux and 10 lux), transmitting wavelengths from 500 to 550 nm. Green light is proven to attract *C.*

caretta and *C. mydas* hatchlings (WITHERINGTON, 1997). In this context it is mentioned that *C. mydas* has best night vision in this range, with maximal rod sensitivity at 502 nm (LIEBMAN and GRANDA, 1971). Hence, dark-adapted hatchlings probably had increased sensitivity for this spectrum, which makes the filter inappropriate for conservation measures.

As outlined above, the Yellow filter (at 21.5 lux) and the Red filter (at 9.6 lux) did not attract *C. caretta* hatchlings to the same extent as the other filters. These findings are consistent with the theory that *C. caretta* shows xanthophobia, which is aversion to yellow light at 560 to 600 nm (WITHERINGTON and BJORNDAL 1991a). However, in my experiments *C. caretta* was not significantly repelled but showed equal distribution in the two-choice box. Therefore, just a limited degree of xanthophobia in hatchlings can be confirmed here. Interestingly, *C. caretta* hatchlings showed circling when tested with the Yellow or the Red filter. Most hatchlings crawled towards the light but changed heading direction later. In comparable experiments with *C. caretta*, circling was observed only when all lights were switched off (WITHERINGTON and BJORNDAL 1991a). Hatchling circling in my experiments may be caused by the shift from dark- to light-adaptation. Light, passing through the Yellow filter ($\lambda > 520$ nm) and the Red filter ($\lambda > 610$ nm), probably first had an attracting effect on my dark-adapted animals, being perceived as brightness rather than as colour cues (rod vision), but later followed by hatchling aversion (xanthophobia) after becoming light-adapted (cone vision). My results suggest that the shift from dark- to light-adaptation in my test animals occurred quickly within a few minutes. Morphological investigation of the retina confirms that the eye of *C. caretta* is adapted for both high spatial resolution and low light sensitivity, but is clearly cone-dominated, with a cone: rod photoreceptor ratio of 2:1 (BARTOL and MUSICK, 2001). In contrast to the *C. caretta* tested, the Yellow filter (7 lux and 21.5 lux) and also the Red filter (4.5 lux and 9.6 lux) highly significantly attracted my *C. mydas* hatchlings. Thus this species showed a clear positive phototaxis for this part of the light spectrum. This is inconsistent with the study conducted by WITHERINGTON and BJORNDAL (1991a), stating that light at 600 nm and 700 nm is relatively unattractive to *C. mydas* hatchlings. Sea turtles' reduced sensitivity to light of higher visible wavelengths > 600 nm is established in the literature (GRANDA and O'SHEA, 1972, WITHERINGTON and MARTIN, 1996, WITHERINGTON, 1997). But newer electrophysiological data suggest that *C. caretta* and *C. mydas* are responsive to wavelengths up to 700 nm (LEVENSON et al., 2004).

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However, the two species tested in my thesis reacted differently to lights in the yellow-red spectrum, whereas both showing attraction to shorter visible wavelengths and UV light. It is suggested here that this could be an adaptation to different habitat use, on land or in water. First, aversion to longer visible-wavelength light could be a protective function for hatchlings emerging during daylight. Though not the common case, diurnal hatchling emergence events are known for both species. At Melbourne Beach, which is a primary *C. caretta* index nesting beach in the U.S., 10% of all emergence events occurred during the afternoon (WITHERINGTON et al., 1990). These hatchlings may be exposed to bright sunlight, which has definite peaks between 500 and 700 nm (green to red). It is under discussion whether sunlight has an effect on the orientation of hatchlings (EHRENFELD and CARR, 1967, VAN RHIJN, 1979, MROSOVSKY, 1970, MROSOVSKY and KINGSMILL, 1985, SALMON and WITHERINGTON, 1995). If so, negative phototaxis for long visible wavelengths between 560 and 600 nm (xanthophobia), as assumed for *C. caretta* (WITHERINGTON and BJORNDAL, 1991a), may be advantageous to avoid possible disorientation during daylight (see Chapter 1, Figure 1.4). However, *C. caretta* does not generally exhibit more diurnal emergence events than *C. mydas*, which shows reduced xanthophobia. At Alagadi Beach, Northern Cyprus, just slightly more *C. caretta* hatchlings emerge during daylight than *C. mydas* (GLEN et al., 2005). Consequently both species face the same selection pressure when emerging in daytime. My observations in Belek, Turkey, also revealed that the sun in fact did attract *C. caretta* hatchlings. If this was due to the peaks in the 500 to 700 nm, or to the UV portion of sunlight, or both spectra, remains unclear. It is suggested here that reduced sensitivity to longer visible wavelength in sea turtles probably plays a minor role on land. More probable is its role in marine habitats. In their first years, *C. caretta* and *C. mydas* hatchlings are found in floating in *Sargassum* rafts, but they undergo a change in habitat when they become juveniles and adults. *C. caretta* are often found in coastal waters within 60 m depth, where they feed on arthropods, mainly crustaceans (SPOTILA, 2004, BJORNDAL, 1997). The proportion of longer-wavelength light is low here, as red, orange and yellow light is absorbed at greater depths. In contrast, *C. mydas* prefer shallower waters inside reefs and bays for feeding (HIRTH, 1997). This benthic habitat provides adequate conditions for the growth of Chlorophyta (chlorophyll b absorption peaks at 454 nm and 643 nm), which is the main component of this species' food. These shallow water habitats allow a good discrimination of colours. Therefore perception of

longer wavelengths at 600 and 700 nm may be of advantage to *C. mydas* rather than to *C. caretta*. The Hawksbill Turtle (*Eretmochelys imbricata*), which is also found in shallow water habitats (coral reefs) feeding on sponges, shows a similar spectral sensitivity to *C. mydas* (WITHERINGTON, 1997). Hence, shallow water habitat-use might explain why *C. mydas* and *E. imbricata* show higher sensitivity to longer visible wavelength than *C. caretta* does. Probably feeding habitats play an important role for the evolutionary development of vision in sea turtles. This is particularly interesting for the Leatherback (*Dermochelys coriacea*), which is known for feeding mainly on jellyfish at remarkable water depths of up to 600 m (ECKERT et al., 1989, HAYS et al., 2004), which is nearly lacking light. The visual capacities of this species in this extreme environment, and a possible shift in sensitivities within life stages as an adaptive mechanism, are relatively unknown to date.

When tested outdoors (Setup 2), hatchling activity was consistently high. Most light-treatment groups were significantly oriented seawards. However, some hatchlings deviated from straight seawards, indicating disorientation (SALMON, 2003). It is suggested here that this was caused by low-degree light pollution from distant settlements visible on the beach, not the previous treatment with experimental light. Evidence for hatchling disorientation was found mainly in dark nights, when no moon was visible, only sky glow. Disorientation by sky glow is also confirmed in the literature (SALMON et al., 1995b) and is also consistent with my findings in Belek. Increased hatchling disorientation was observed during dark nights (Chapter 4). However, increased exposure time to artificial lights is known to affect the subsequent sea-finding ability of hatchlings. LORNE and SALMON (2007) found that a short (two minutes) landward crawl towards artificial lights had no effect on the sea-finding ability of *C. caretta* hatchlings. In contrast, hatchlings which were exposed to artificial lights and crawled in this direction for over two hours were not oriented sea-wards subsequently. My test animals were exposed to artificial lights set landwards only for a maximum five minutes, which probably did not interfere with their later sea-finding ability. Moreover, my hatchlings were in varying stages of dark-adaptation, ranging from one minute to over 60 minutes, depending on the time spent in the covered buckets after the light treatments (Setup 1). No difference in orientation behaviour in these hatchlings was observed on the beach (Setup 2), supporting a rapid dark-adaptation in all test animals. This is within the range of values in the literature available for freshwater turtles *Pseudemys scripta*, with 90 seconds

required for dark-adaptation after treatment with red light (GRANDA et al., 1972) or a tungsten lamp of white light (ZWICK and GRANDA, 1968). Similarly, hatchlings tested for sounds only (Setup 1), which were all completely dark-adapted, also showed significant seaward orientation on the beach. Consequently, turning off artificial lights at nesting beaches immediately after an observed hatchling emergence event may have a positive effect on disoriented hatchlings. However, this must be considered a "last minute" measure. A quick-moving, disoriented *C. mydas* hatchling will be lost between vegetation and dunes much faster than a slower *C. caretta* hatchling, with a limited possibility of restoring sea-finding. STAPPUT and WILTSCHKO (2005) found that hatchlings that had previously crawled 5 m on the beach were able to find the sea in total darkness, whereas hatchlings that were denied crawling were disoriented in the dark. The authors concluded that a short crawl on the beach could set the magnetic compass course while still on land. This proposed mechanism highlights a problem for hatchlings being disoriented by artificial lights, as turning off artificial lights during the crawl would not have a corrective effect on them. Based on my findings, there is a need for conservation measures also in Belek, which will be described in Chapter 6.

Sounds and lights:

Combined testing of lights and wave/surf sounds confirmed that light plays the crucial role in providing orientation cues to hatchlings. In most cases hatchlings were significantly oriented towards the lights, but not the wave/surf sounds. Inhibition of movements was not observed in most of these treatment groups. *C. caretta* hatchlings were randomly distributed in the experimental box when tested for the Yellow or Red filter light and wave/surf sounds. Based on the outcomes of the previous tests, using sounds or lights exclusively, it is concluded here that these results are not caused by an attraction towards wave/surf sounds but indifference or aversion to yellow light (xanthophobia). In contrast to the previous setup testing lights solely, *C. mydas* showed increased aversion to the Yellow filter (7 lux) in presence of wave/surf sounds. Given that wave/surf sounds do not have an attracting effect, this is evidence for a certain degree of xanthophobia also in this species, which was not highlighted in the literature before (WITHERINGTON and BJORNDAL, 1991a).

Conclusions

My findings allow conclusions on the behaviour and sensory abilities of the two sea turtle species. Based on overall agreement with the previous study of WITHERINGTON and BJORNDAL (1991a), who conducted indoor tests near Melbourne Beach, Florida (see Fig. 3.5), it is concluded here that my behavioural data obtained at Xcacel in Mexico are representative also for other populations, namely the ones in the Mediterranean. The ability to detect a wide spectrum of light, including UV light, confirms the important role played by vision in sea turtles, and highlights the need to differentiate between human and turtle vision capabilities when assessing light pollution on nesting beaches. In my experiments *C. caretta* and *C. mydas* hatchlings both showed high attraction to short visible wavelengths. Additionally, I confirmed xanthophobia in *C. caretta* and found evidence also in *C. mydas*. Reducing illuminance levels at nesting beaches alone may not be appropriate to reduce hatchling disorientation. This is, as hatchlings showed higher attraction to a UV-transmitting light source with lower illuminance (2.1 lux and 3.8 lux) compared with the Yellow filter (7 lux and 21.5 lux) and Red filter (4.5 lux and 9.6 lux). Attaching yellow and red dichroic filters to a polychromatic light source was found to be adequate to mitigate disorientation in *C. caretta*, but found to be less effective in *C. mydas* hatchlings. This is a possible approach for conservation measures to reduce hatchling disorientation mainly on *C. caretta* nesting beaches. Further research in the field is recommended to investigate whether yellow or red filters attached in front of high-intensity stationary light sources, such as Metal Halide lamps, can in fact significantly deter hatchlings from landward crawls. This may be of importance at nesting sites where artificial lights are maintained, but where there are limited possibilities to shield them or switch them off, e.g. at beaches affected by mass tourism (see Chapter 4). It is pointed out that a filter cutting out $\lambda < 520$ nm has a similar effect as a Low Pressure Sodium Vapour (LPS) light and will probably have a lesser impact on female nesting, compared with a High Pressure Sodium Vapour (HPS) light with peaks at 575 nm to 625 nm but also in the shorter spectrum (see Chapter 1, Table 1.3). The negative effect of the latter source was observed in Belek, where females avoided nesting along a coastal street close to HPS lights, operated at night (Chapter 4). Switching off lights a reasonable time after an emergence event may probably be considered the ultimate measure to restore sea-finding in disoriented hatchlings. But regarding long-term sea turtle conservation, it is

recommended that a strong focus should be made on controlling overall light pollution on nesting beaches.

The results of this thesis indicate that although turtles are able to perceive low-frequency sounds at moderate intensities, seaward orientation on land in the presence of acoustic cues is not confirmed, at least for hatchlings. Hatchlings tested in indoor experiments did not show significant orientation towards wave/surf sounds. Artificial low-frequency sounds did not elicit a significant aversion in hatchlings, whereas there was evidence for inhibition of crawling behaviour. This conclusion is mainly based on the high inactivity observed in the sound-treatment group, which was contrary to the normal activity (frenzy) on the beach in the same treatment groups. Experimental wave/surf sounds, including substrate vibrations, did not mitigate the negative effect that lights had on the orientation of hatchlings. Except for the Yellow and Red filters, hatchlings always chose the light over the sound source/vibrations (Fig 5.14 a, b). Thus the preliminary results of my experimental setup on a small sample size of *C. caretta* hatchlings in Belek (Chapter 4.2.4) were qualified as not representative. However, as visual cues are unreliable under natural conditions, e.g. during overcast conditions (see Chapter 1, Figure 1.6), visual orientation may not be exclusive. For sea-finding, a combined mechanism, including a magnetic compass (LOHMANN et al., 1995, STAPPUT and WILTSCHKO, 2005), olfactory (GRASSMAN et al., 1984, GRASSMAN and OWENS, 1987) and auditory perception (LENHARDT et al., 1983) may have evolved.

Caretta caretta tested for lights

Caretta caretta tested for sounds

Chelonia mydas tested for lights

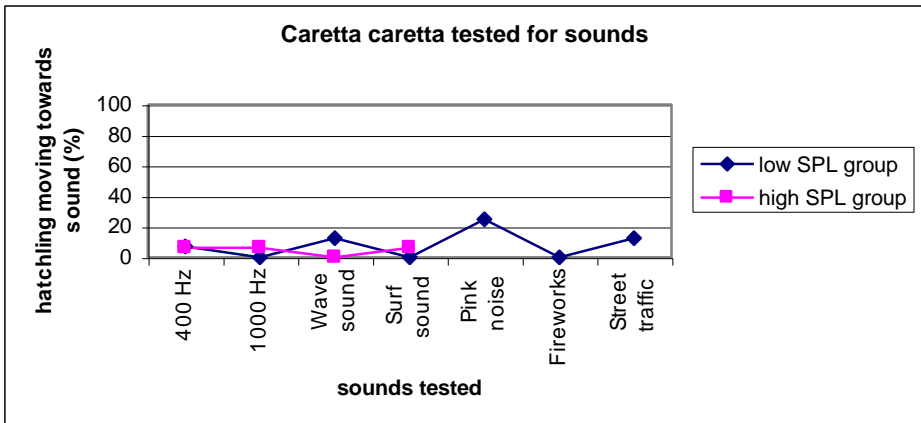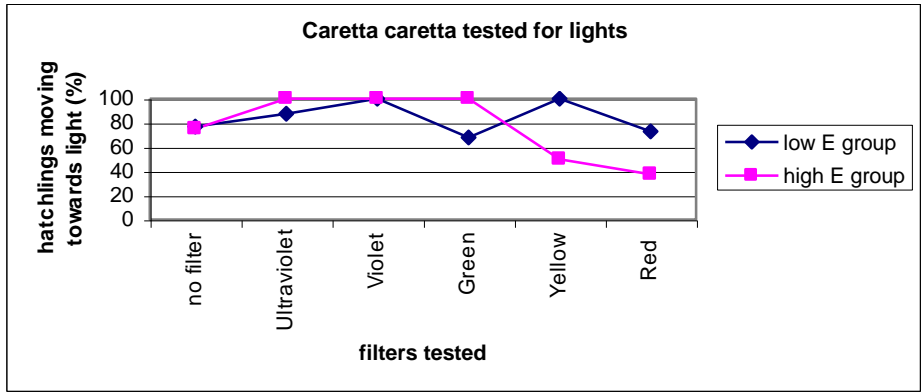**Chelonia mydas tested for sounds**
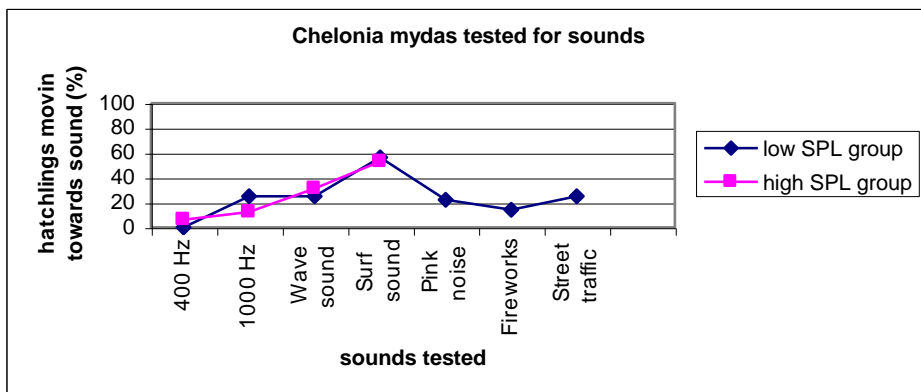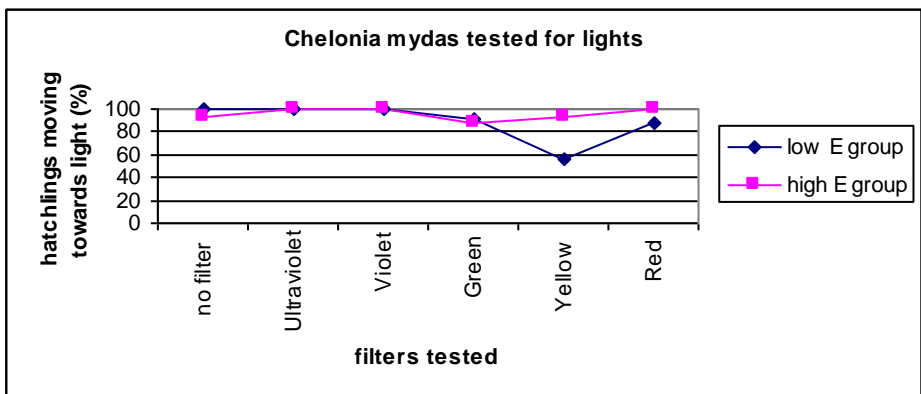
Figure 5.18: Percentage of hatchlings moving towards a provided stimulus (sound or light) in experiment. Note that both species generally show preference for lights, but this is decreased for the Yellow (< 520 nm cut off) and Red (< 610 nm cut off) filter attached in front of the Tungsten Halogen lamp. In contrast, generally both species are indifferent to sound or avoid the direction of the sound stimuli, indicating a possible adverse effect on hatchlings.
E (Illuminance) and SPL (Sound Pressure Level) measured according to Tables 5.2 and 5.3.

6 Recommendations for sea turtle conservation at nesting beaches in the Mediterranean

Coastal development is assessed as one of the main threats for sea turtles by the Marine Turtle Specialist Group (MTSG, 2007; see Chapter 1). In the eastern Mediterranean, mass tourism is a major cause of coastal development, particularly in Greece and Turkey, which hold important *C. caretta* and *C. mydas* nesting sites (VENIZELOS, 2001, CANBOLAT, 2004). As it contributes to economic development, mass tourism is strongly supported by the local governments. However, in the long term it will lead to a degradation of natural resources and a reduction of biodiversity in the Mediterranean. The results of my thesis confirm that coastal development and its associated light pollution have a negative impact on the nest site selection of female sea turtles and hatchling seaward orientation. Females avoid beaches with intensive artificial lighting in front of hotels, on other recreational infrastructure, and along coastal streets. This is probably contributing to a long-term reduction in nesting success in areas that traditionally provided suitable nesting grounds, but are nowadays subject to coastal development. Hatchlings are affected by the disorientation caused by polychromatic Metal Halide and also High Pressure Sodium Vapour lights, which are associated with hotel zones and coastal streets. Hatchling disorientation has also increased in undeveloped areas, which are preferred by females over these brighter illuminated hotel zones. The problem of light-trapping in these split-up nesting niches between hotel zones was demonstrated in Belek, Turkey. I showed that hatchling disorientation is correlated with increased hatchling mortality in Belek, and it must be assumed that the effective hatchling mortality caused by light pollution is even higher due to unreported cases. Moreover, it cannot be excluded that disorientation on land also contributes to higher mortality at sea. This is because long crawls on the beach weaken the hatchlings and reduce their overall fitness, which is essential for escaping predators and swimming further offshore to their foraging grounds. Mediterranean sea turtle nests generally show low hatching success and high predation (FRAZIER, 2001). Also taking into account hatchlings' low survival rates, which may be only 1-2 of 1000 hatchlings reaching maturity (FRAZER, 1986), and low reproduction rates in adult sea turtles (LIMPUS and CHALOUPKA, 1997), the anthropogenic factors contributing to the reduction of the survival chances of these endangered species in their critical habitats, the nesting beaches, must be eradicated.

Therefore, conservation priorities and actions in Belek, Turkey, and throughout the Mediterranean, should include:

Reducing mass tourism at sea turtle key nesting sites

Mass tourism is evident at a number of nesting sites in Greece and Turkey (Table 6.1), but is also threatening potential nesting areas which are only moderately frequented by tourists at present, such as the Sidero Peninsula on Crete[24].

Table 6.1: Mediterranean index nesting sites affected by mass tourism (2005)

Country	Nesting site
Turkey	Belek, Side, Kizilot, Dalyan, Fethiye, Kumluca, Demirtas, Anamur, Sogözü
Greece	Zakynthos, Kefalonia, Chania, Rethymno, Messara Bay
Cyprus	Akamas Peninsula, Lara/Toxeftra

For conserving sea turtle index nesting sites in the Mediterranean, the primary goal is to prohibit a further extension of mass tourism here, under governmental law. Instead, a <u>sustainable tourism or ecotourism,</u> with strict limitation of new permissions for construction sites and hotel complexes by law, should be aimed for. This should also include a governmental decree for prohibiting the construction of new hotels near to the shoreline (< 1 km) at index nesting sites.

Mitigation measures on a local scale

Besides preventing mass tourism, special mitigation measures have to be considered for nesting areas which are already affected by coastal development. The majority of hotels and construction sites in the Tourism Development Area of Belek, Turkey (Section II, see Chapter 4), are located within 100 m of the shoreline. This is a considerable problem for the sea turtles here. Dredging and sand removal for construction is destroying nesting space (Figures 4.10 a, b). Another problem is sand compaction by cars and construction vehicles (Figure 4.9). In front of hotels, the presence of humans on the beach is disturbing nesting females (Figure 4.8).
In Belek, tourists were informed about the nesting females by the team from Hacettepe University, who did the sea turtle monitoring in 2005. Tourists were asked to avoid nesting beaches at night. Though some tourists were insightful, this task is

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[24] www.minoangroup.com

difficult to achieve in designated hotel zones, as the beach at night is one of the main attractions to tourists. Further awareness training and continuous cooperation with local research and conservation groups is recommended here. In Belek, considerable noise pollution was present in the *C. caretta* egg-laying zone, which probably contributes to deterring females. A primary mitigation measure is reducing high intensity noise after nightfall here. Unfortunately, this is again interfering with the interests of hotel owners and visitors, since discotheques, fireworks, etc. are part of the recreational activities in the area. Thus there is only a slight chance for governmental legislation for these measures and their realization will depend on the decisions of single hotel owners. As sound levels attenuate over distance, it is also strongly recommended to relocate the dance floors that were observed in Belek close to the shore. This will contribute to reducing overall noise pollution at the nesting beach to below a level which is sustainable for nesting females. It is hardly possible to introduce mitigation measures for reducing road traffic noise, which was present along the coastal road in Section II at 65 m from the shore and 30 m from sea turtle nests. The only measure is overall reduction of road traffic during the nesting season, which will be difficult to achieve during tourism peak times.

The main problem associated with hotel zones in Belek is light pollution from polychromatic light sources, which is affecting the nearby egg-laying zone. This makes mitigation measures difficult here. Therefore, a governmental decree for reducing overall light pollution levels should be aimed for. For new hotels planned, this must include the regulation for a minimum distance of 1 km from the shoreline. Only this will enable the implementation of light barriers, such as sand dunes and Mediterranean sclerophyllous evergreens between the hotels and the seashore. Artificial sand dunes at construction sites were shown to block direct irradiation from close light sources in Belek (see Appendix 3 d, e). However, artificial sand dunes are not acceptable within the egg-laying zone (< 35 m from the shore), as they reduce nesting space (see Figs. 4.10 a, b). Moreover, they have a limited mitigating effect against sky glow, which was also present in Belek. It is questionable if the measures proposed here are feasible at all. There is an economic risk that tourists will reject hotels which are not built close to the seashore. Nevertheless, with regard to sea turtle conservation, a buffer zone between hotels and the egg-laying zone must be considered the primary mitigation measure for sea turtle index nesting sites.

Further steps for reducing the negative impact of light pollution on sea turtles should include:

Switching off lights close to sea turtle nests:

This measure is not feasible for the entire nesting area, due to the multi-origin of light sources from hotels, bars and other recreational facilities. But it can be considered as a local measure, which must be discussed with the hotel staff. Switching off a strong light source in direct proximity of a nest, from which hatchling emergence is expected, was shown to reduce hatchling disorientation in Belek.

Light shielding:

This is the first alternative for lights kept operating the entire night. High Pressure Sodium Vapour (HPS) lights on the coastal road in Section II, Belek, were found to have poor shielding, resulting in a considerable illumination of the nesting beach. Appropriate fixtures avoiding light trespass are recommended here. For Metal Halide (MH) lights, shielding is not desired by hotel owners, as these sources are used as floodlights for night work on the beach and for security reasons at hotels. Alternative mitigation measures are recommended below.

Shielding nests:

Shielding nests may be feasible for single cases in proximity to hotel zones, but is difficult to handle for the entire nesting area. However, fieldwork in Belek showed that shielding a single nest with a curtain against a MH light reduced hatchling disorientation. This measure is also dependent on the cooperation of hotel owners.

Replacing polychromatic light sources:

Replacement of High Pressure Sodium Vapour (HPS) with Low Pressure Sodium Vapour (LPS) lights having monochromatic properties was a crucial step in reducing hatchling disorientation at index nesting sites in the United States (WITHERINGTON and MARTIN, 1996, Florida Power and Light Company, 2002; see Chapter 1). The implementation of LPS lights in the Mediterranean is difficult for various reasons. First, LPS lamps are not used on a routine basis in most parts of Europe due to higher maintenance costs compared with HPS or Mercury Vapour lamps (TERSALVI, NEUGEBAUER, pers. comm.). Second, these lamps can be used exclusively for sites in which the light colour is secondary, which limits their application. At least for coastal roads at index sites, these LPS lamps would be an alternative to HPS lamps. Though more expensive, the luminous efficiency of LPS lights is higher, with up to 200

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lm/watt compared with HPS lights (150 lm/ watt), which may compensate for higher maintenance costs. However, the implementation of LPS at nesting sites will mainly depend on technical feasibility, governmental decrees, and financing. For polychromatic Metal Halide lights, which contributed to the highest illuminance (up to 40 lux) measured in the egg-laying zone of Belek, there is no cheaper technical alternative. These lights have low maintenance costs, a good colour rendering, and a luminous efficiency of 80 lm per watt. This makes them the primary choice in front of hotels, which need to be lit up for reasons of security as well as of cleaning and working operations. Therefore mitigation measures could include the stricter regulation of these sources' nighttime activity, e.g. a shut-off at 9-10 pm at the latest, which is the onset of female nesting and hatchling emergence at night. Another option is using colour filters attached in front of the lamp.

Using colour filters:

In Xcacel, Mexico, I experimentally tested five dichroic filters attached in front of a polychromatic light source (see Chapter 5). Two filters were found adequate to significantly reduce sea turtle hatchling disorientation. These were the Yellow filter, cutting off $\lambda < 520$ nm and the Red filter, cutting off $\lambda < 610$ nm. Overall, *C. caretta* showed lower disorientation if tested with these filters compared with *C. mydas*. Therefore these dichroic filters are recommended as a mitigation measure, particularly for the *C. caretta* nesting site in Belek. According to TERSALVI (pers. comm.) the dichroic filters used in my experiments are technically suitable for fixture on High Pressure Sodium Vapour (HPS) lamps and also on high wattage Metal Halide (MH) lamps, due to their heat resistance. The minimum cost proposal for the Yellow filter (code: LW520) of 11 cm x 16 cm size, which is one of the competitive models manufactured by ODL, is about 22 Euros, if purchased in larger quantities. Hence for usage in front of HPS lights, the Yellow filter is an economic alternative to the expensive replacement with monochromatic Low Pressure Sodium Vapour (LPS) lights. HPS lights have distinctive peaks at 570 and 630 nm (see Table 1.3). The Yellow filter cuts off $\lambda < 520$ nm; hence it does not considerably bias the HPS lamp's visible colour (Fig. 6.1). This filter also excludes wavelengths in the UV range (< 380 nm), which is not visible to humans but attracts sea turtle hatchlings (see Fig. 5.8 e). Consequently, the Yellow filter will not affect colour perception in humans to an intolerable extent (NEUGEBAUER, pers. comm.), but may significantly reduce *C. caretta* hatchling disorientation on the beach (see Figure 5.13 a).
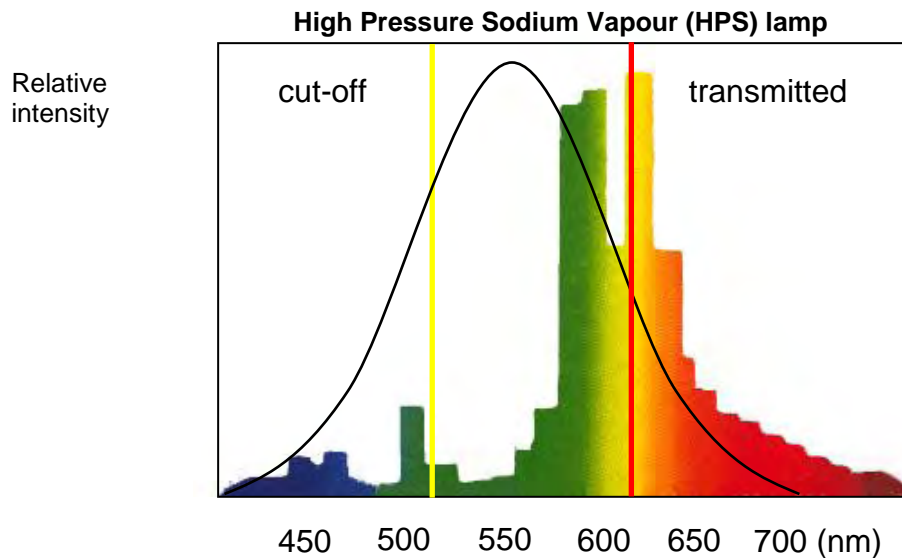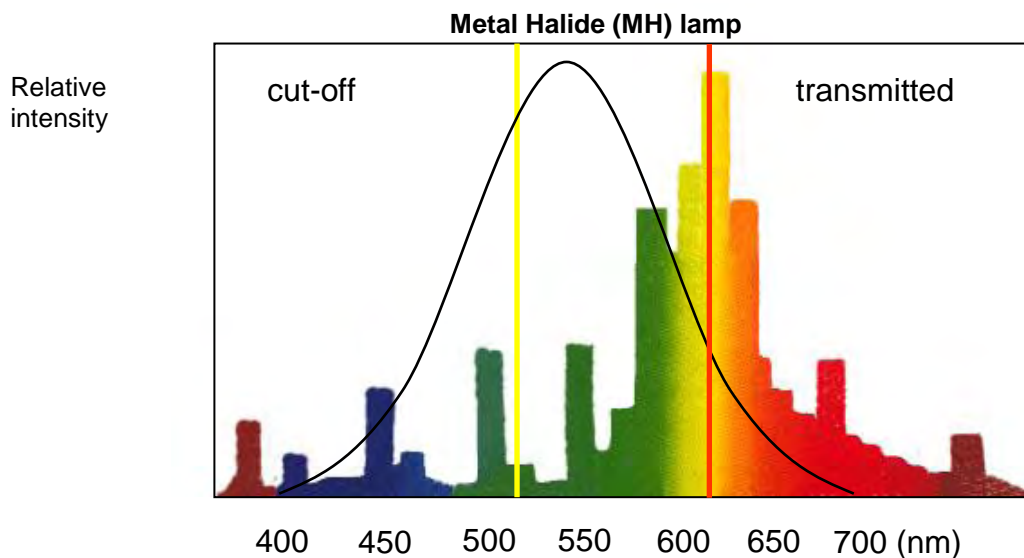
Figure 6.1: The Yellow colour filter cuts off the spectral emission of a Lucalox HPS lamp at < 520 nm (yellow line). The Red colour filter cuts off a higher portion of the emitted light < 610 nm (red line). The human eye sensitivity for photopic adaptation (peak at green/yellow) is indicated by the V (λ) curve.

For use on polychromatic Metal Halide lights, the Yellow filter is also technically applicable, but it is pointed out that the lamp's visible colour for humans will be considerably changed when used with this filter. The Yellow filter cuts off the lamp's violet, blue, and a proportion of the green spectrum, and shifts the white colour of this broadband source to a perceived yellow (Figure. 6.2). Consequently, it also significantly reduces the effective illuminance on the beach perceived by humans (Table 6.2). For sea turtles this is desirable, but is not wanted on beaches, which require high illumination levels and good colour rendering for night work (Chapter 4).

Figure 6.2: The Yellow colour filter cuts off the spectral emission of a Metal Halide (MH) lamp at < 520 nm (yellow line). The Red colour filter cuts off at < 610 nm (red line). The human eye sensitivity for photopic adaptation (peak at green/yellow) is indicated by the V (λ) curve. Note that the MH light source emits a higher proportion of visible light < 500 nm compared to the HPS lamp (Figure 6.1). Therefore these filters also change the MH lamp`s colour and also considerably reduce the illuminance perceived by humans.

Table 6.2: Change in overall illuminance using colour filters attached in front of a Metal Halide lamp[1]

Metal Halide lamp (70 watts)	Wavelength (λ) cutoff	Illuminance E_V (at 1m)
Unfiltered light	full spectrum (350-850 nm)	3500 lux
Yellow colour filter (manufacturing code: LW520)	< 520 nm	2100 lux
Red colour filter (manufacturing code: LW610)	< 610 nm	150 lux

[1] At 1 m distance from the source. Note that the Red filter cuts off a larger proportion of the emitted light compared to the Yellow filter, which results in lower Illuminance E_V.

While developing mitigation measures, one must keep in mind that yellow filters attached to polychromatic light sources will not only reduce hatchling disorientation by eliciting xanthophobia (WITHERINGTON and BJORNDAL, 1991b) but still may deter females from the nesting beaches.

Relocating nests from areas of light pollution:

This can be considered a preventive measure for areas that are seriously affected by light pollution, e.g. in hotel zones. Relocation to a hatchery may reduce the mortality risk caused by artificial lights. However in Belek this measure must be used in combination with light shielding, because even darker areas (see Chapter 4, Appendix 3) are affected by long-distance light propagation. One counterargument for nest relocation is increased hatchling mortality due to predators in undeveloped areas (see Chapter 4). This may be counterproductive, regarding the conservation measures aimed for. Further studies investigating hatchling survival in relocated nests in Belek are recommended here.

Due to the high degree of coastal development at the beaches on the Mediterranean, hatchling conservation must be one of the primary goals. But there are a number of other anthropogenic threats affecting sea turtles, in particular the post-hatchling life stages and adults. In the Mediterranean this is, in the first instance, the problem of bycatch (see chapter 2.0). According to the European Commission Project 98/008 (LAURENT et al., 2001) fishery activity is considered the most important anthropogenic mortality factor known for the endangered *C. caretta* and *C. mydas* breeding in the Mediterranean. Continuous reduction of mature females and males will inevitably reduce hatchling output in these long-lived but slow-maturing species. Thus, along with conserving the nesting beaches, it is crucial to further concentrate on the conservation of large juveniles and adults in their marine habitats. Save the adults if you want hatchlings.

Acknowledgements

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Erklärung

Hiermit erkläre ich, die vorliegende Arbeit selbstständig verfasst, keine anderen als die angebebenen Quellen und Hilfsmittel benutzt und die Zitate kenntlich gemacht zu haben. Die vorliegende Arbeit wurde weder von mir noch von sonstigen Personen anderweitig als Dissertation eingereicht. Es gab keine früheren Promotionsversuche.

APPENDIX 1 a

Top ten *Caretta caretta* global nesting sites (per countries). Conversions in *italic*, categories (A-E) as defined in Table 2.2

Caretta caretta	latitude	longitude	nests (min)	nests (max)	females (min)	females (max)	year	source
Oman						**30000 (A)**		
Masirah Island and Kuria Muria Island	20,4167	58,8333				30000	1979	Ross (1979)
Masirah Island	20,4167	58,8333			10000	100000	2005	UNEP-WCMC (2005)
Masirah Island	20,4167	58,8333				30000	2005	SWOT (2007)
USA						**20000-30000 (A)**		
Melbourne Beach, Florida	28,0257	-80,5538	37242	68614	9083	16735	1988-1992	Meylan et al. (1995)
South Florida	27,75	-80,3833	48531	83442	*12133*	*20861*	1989-1998	TEWG (2000)
Hutchinson Island, Florida	27,3627	-80,2342	3121	8214	*780*	*2054*	1981-1998	TEWG (2000)
Northwest Florida (Panhandle)	30,4833	-86,5	1200			*300*	1998	TEWG (2000)
Northeast Florida-North Carolina	33,0333	-79,3333	7500			*1875*	1998	TEWG (2000)
Cape Island, South Carolina	33,0488	-79,3188	579	2654	*145*	*664*	1975-1998	TEWG (2000)
Melbourne-Wabasso Beach, Florida	27,8341	-80,4356				36364	1989-1999	Weishampel et al. (2003)
Florida				52469		*13117*	2005	SWOT (2007)
South Carolina				4078		*1020*	2005	SWOT (2007)
Georgia				1199		*300*	2005	SWOT (2007)
North Florida				647		*162*	2005	SWOT (2007)
Australia						**5000-6000 (B)**		
Western Australia (total)						5000	2004	Limpus and Chatto (2004)
Dirk Hartog Islands	-25,75	114			1000	5000	1999	UNEP-WCMC (2005)
Murion Islands	-21,6965	114,311			100	500	1999	UNEP-WCMC (2005)
Cape Range, Ningaloo	-21,949	113,948			100	500	1999	UNEP-WCMC (2005)
Shark Bay and Dirk Hartog Islands	-25,795	113,689				no data	2005	SWOT (2007)
Bungelup						659	2005	SWOT (2007)
Northwest Cape					150	200	2005	SWOT (2007)
Baleman's Beach, Coral Bay Division						39	2005	SWOT (2007)
Eastern Australia (S-Queensland)						3500	1970	Limpus and Chatto (2004)
Eastern Australia (S-Queensland)						500	2003	Limpus and Chatto (2004)
Mon Repos	-24,8091	152,455			100	500	1998	UNEP-WCMC (2005), Hepell et al. (1996)
Swain Reef (combined)	-21,92	152,517			100	500	1998	UNEP-WCMC (2005)
Wreck Island	-23,3377	151,96				1000	1979	Limpus, pers. Com. in Ross (1979)
Capricorn and Bunker Islands	-23,4621	152,025				1000	1972	Bustard (1972) in Ross (1979)
Wreck Rock Beach			120			62	2005/2006	SWOT (2007)

Caretta caretta	latitude	longitude	nests (min)	nests (max)	females (min)	females (max)	year	source
Cape Verde Islands						3000-4000 (C)		
Boa Vista Island, Cape Verde	16,2488	-22,5718	863	2619			2001-2004	Varo et al. (2005)
Boa Vista Island, Cape Verde	16,2488	-22,5718				4000	1998-2004	Varo et al. (2005)
Cape Verde (total)				5396		3121	2005	SWOT (2007)
Brazil						1000-2000 (C)		
Espirito Santo (state)	-19,8221	-39,9848		736		*184*	1991-1997	Baptistotte et al. (2003)
Praia do Forte, Bahia	-13,5373	-38,9059		322		*81*	1987-1993	Marcovaldi and Laurent (1996)
Maranhao and Ceara	-4,51863	-37,6648		1000		*250*	1990	Marquez (1990)
Brazil (total)				5285		*1321*	2005	SWOT (2007)
Greece and Turkey						2000-3000 (C)		
Mediterranean (total)					2280	2787	1995-2000	Broderick et al. (2002)
Zakynthos, Greece	37,6731	20,9151		2000		*500*	1990	Marquez (1990)
Laganas Bay, Zakynthos, Greece	37,6731	20,9151		1294		*324*	1984-2002	Margaritoulis (2005)
Kyparissia Bay, Greece	37,2773	37,2773		620		*155*	1990-2000	Margaritoulis and Rees (2001)
Greece (total)				1384		*346*	2005	SWOT (2007)
Southern Cyprus (total)	34,91	32,32		332		*83*	2005	SWOT (2007)
Belek, Turkey	36,8536	31,0417		647		*162*	1999-2000	Canbolat (2001)
Turkey (total)				1125		*282*	2005	SWOT (2007)
Northern Cyprus (total)	35,3	33,5		234		*59*	2005	SWOT (2007)
South Africa						< 500 (E)		
Tongaland	-28,5597	32,234				500	1974	Hughes (1974) in Ross (1979)
Sodwana	-28	32,45			600	1000	1963-1997	UNEP-WCMC (2005)
South Africa (total)				1728		238	2005	SWOT (2007)
Japan						< 500 (E)		
Maehama, Yakushima Islands	30,3881	130,411			100	500	1999	UNEP-WCMC (2005)
Inakahama, Yakushima Islands	30,3881	130,411			100	500	1999	UNEP-WCMC (2005)
Japan (Yakushima)	30,3881	130,411		1758		*440*	2005	SWOT (2007)
Mexico						< 500 (E)		
Quintana Roo, Mexico	20,3387	-87,3485				500	1976	Marquez (1976) in Ross (1979)
Eastern Yucatan Peninsula	20,3387	-87,3485	1500	2300	*375*	575	1998	TEWG (1998)
Yucatan Peninsula, Quintana Roo	20,3387	-87,3485		1052		*263*	1998	TEWG (2000)
Mexico (Quintana Roo total)				961		*240*	2005	SWOT (2007)

APPENDIX 1 b   Top ten *Chelonia mydas* global nesting sites (per countries). Conversions in *italic*, categories (A-E) as defined in Table 2.2

Chelonia mydas	latitude	longitude	eggs	nests (min)	nests (max)	females (min)	females (max)	year	source
Australia						?	?		
Western Australia						?	?		
Barrow Island	-20,75	115,417				10000	100000	1999	UNEP-WCMC (2005)
Lacepede Islands	-16,867	122,183				10000	100000	1999	UNEP-WCMC (2005)
Murion Islands	-21,6965	114,311				1000	5000	2005	UNEP-WCMC (2005)
Dampier Archipellago	-20,5986	116,746				1000	5000	2005	UNEP-WCMC (2005)
Cape Range, Ningaloo	-22,3224	113,915				1000	5000	2005	UNEP-WCMC (2005)
Eastern Australia (Queensland)							40000 (A)		
Raine Island	-11,6	144,02					40000	2005	Limpus and Limpus (2005)
Raine Island	-11,6	144,02					11538	1974-1979	Seminoff (2004)
Raine Island	-11,6	144,02					18000	2001	Seminoff (2004)
Heron Island	-23,4592	151,93					400	1964-1969	Seminoff (2004)
Heron Island	-23,4592	151,93					562	1993-1999	Seminoff (2004)
Moulter Cay	-11,45	144				5000	10000	1998	Limpus et al (2003)
Raine Island	-11,6	144,02				5000	10000	1998	Limpus et al (2003)
Bountiful Islands	-16,6641	139,867				1000	5000	2005	UNEP-WCMC (2005)
Murray Islands	-9,92	144,054				1000	5000	2005	UNEP-WCMC (2005)
Pisonia Islands	-16,4923	139,809				1000	5000	2005	UNEP-WCMC (2005)
Sandbank	-13,3827	143,974				1000	5000	2005	UNEP-WCMC (2005)
Capricornia, Heron Islands	-23,4592	151,93				1000	5000	2002	Limpus et al (2003)
Costa Rica							20000-30000 (A)		
Tortuguero	10,536	-83,4941			104411	17402	37290	1999–2003	Troëng, Rankin (2005)
Tortuguero	10,536	-83,4941					8333	1975	Seminoff (2004)
Tortuguero	10,536	-83,4941				21952	23522	2002	Seminoff (2004)
Oman							6000 (B)		
Ras Al Hadd	22,5333	59,8				6000	18000	1991	Salm (1991)
Ras Al Hadd	22,5333	59,8				10000	100000	2005	UNEP-WCMC (2005)
Ras Al Hadd	22,5333	59,8					6000	1979	Seminoff (2004)
Ras Al Hadd	22,5333	59,8					6000	1988	Seminoff (2004)
Comoros Islands							5000 (B-C)		
Comoros Islands	-11,9619	43,9134					1850	1973	Seminoff (2004)
Comoros Islands	-11,9619	43,9134					5000	2000	Seminoff (2004)

Chelonia mydas	latitude	longitude	eggs	nests (min)	nests (max)	females (min)	females (max)	year	source
Indonesia							**< 5000 (B-C)**		
Indonesia (total)				10000	20000	*3333*	*6666*	2001	Halim et al. (2001)
Sangalaki, Berau Islands	2,30144	118,504			4911		*1637*	2004	Meyer and Zindel (pers. comm.)
Sangalaki, Berau Islands	2,30144	118,504			5065	1000	5000	2000	UNEP-WCMC (2005)
Pulau Bilang-Bilangan, Berau	2,30144	118,504				1000	5000	1990	UNEP-WCMC (2005)
Berau Islands	2,30144	118,504					36000	1940's	Seminoff (2004)
Berau Islands	2,30144	118,504					4500	1984	Seminoff (2004)
Pangumbahan (West Java)	-6,44358	105,661					8333	1950's	Seminoff (2004)
Pangumbahan (West Java)	-6,44358	105,661					1333	1980's	Seminoff (2004)
Suka Made (East Java)	-7,66713	114,459					518	1970-1974	Seminoff (2004)
Suka Made (East Java)	-7,66713	114,459					132	1991-1995	Seminoff (2004)
Seychelles							**< 5000 (B-C)**		
Aldabra Atoll	-9,44259	46,3433				1000	5000	2005	UNEP-WCMC (2005)
Seychelles	-9,44259	46,3433					12000	1900	Seminoff (2004)
Seychelles	-9,44259	46,3433					4145	1996	Seminoff (2004)
St. Helena, United Kingdom							**4000-5000 (C)**		
Ascension Island	-7,94916	-14,3596			13881		*4627*	1998-1999	Godley et al. (2001)
Ascension Island	-7,94916	-14,3596				1542	2314	1970's	Godley et al. (2001)
Ascension Island	-7,94916	-14,3596					2670	1978	Seminoff (2004)
Ascension Island	-7,94916	-14,3596					3709	2001	Seminoff (2004)
Malaysia							**4000-5000 (C)**		
Sabah Turtle Islands (combined)	6,17	118,079			7156	1000	5000	2001	UNEP-WCMC (2005)
Ti Bakkungan Kechil, Sabah	6,17	118,079	303919		3385		1199	1997	UNEP-WCMC (2005)
Ti Selingan, Sabah	6,17	118,079	445477		5189		2216	1997	UNEP-WCMC (2005)
Ti Gulisan, Sabah	6,17	118,079	233431		2618		867	1997	UNEP-WCMC (2005)
Sabah	6,17	118,079					1854	1965-1968	Seminoff (2004)
Sabah	6,17	118,079					3251	1995-1999	Seminoff (2004)
Sarawak	3,04059	112,605					7549	1927-1934	Seminoff (2004)
Sarawak	3,04059	112,605					762	1998-1999	Seminoff (2004)
Peninsular Malaysia	2,5293	104,059					3096	1961	Seminoff (2004)
Peninsular Malaysia	2,5293	104,059					1057	1993	Seminoff (2004)

Chelonia mydas	latitude	longitude	eggs	nests (min)	nests (max)	females (min)	females (max)	year	source
Phillipines							**2000-3000 (C)**		
Tawi-Tawi Turtle Islands (total)	5,09773	120,15					1000	1984-1989	Trono (1991)
Tawi-Tawi Turtle Islands (total)	5,09773	120,15	1121233		*9750*		*2437*	1984-1989	Trono (1991)
Ti Baguan	-0,1019	118,313	1172610			1000	5000	1995	UNEP-WCMC (2005)
Ti Taganak	-0,1019	118,313	241 033		3038	600	1000	1995	UNEP-WCMC (2005)
Phillipines	5,09773	120,15					4886	1951	Seminoff (2004)
Phillipines	5,09773	120,15					3198	1981-1985	Seminoff (2004)
Brazil							**3000 (C)**		
Isla Trinidade	-20,5142	-29,3275					3000	1981	Seminoff (2002)
Isla Trinidade	-20,5142	-29,3275					3000	2000	Seminoff (2002)

Nests evaluated in monitoring area AY (Section I Belek) in 2005

nest ID	latitude/longitude	total tracks	oriented	disoriented	reached sea	died
AY61	36 51 454 N, 30 58 321 E	no data				
AY36	36 51 456 N, 30 58 225 E	no data				
AY43	36 51 455 N, 30 58 196 E	43	37	6	43	0
AY5	36 51 461 N, 30 58 195 E	25	25	0	25	0
AY42	36 51 430 N, 30 57 829 E	30	20	10	14	16
AY123	36 51 424 N, 30 57 718 E	no data				
AY10	36 51 380 N, 30 57 164 E	32	30	2	13	19
AY17	36 51 305 N, 30 57 046 E	6	5	1	6	0
AY73	36 51 243 N, 30 55 450 E	45	18	27	45	0
AY47	36 51 353 N, 30 56 914 E	44	35	9	44	0
AY72	36 51 354 N, 30 56 897 E	32	30	2	29	3
AY162	36 51 349 N, 30 56 870 E	45	30	15	45	0
AY135	36 51 351 N, 30 56 868 E	38	31	7	27	11
AY108	36 51 351 N, 30 56 824 E	47	41	6	47	0
AY118	36 51 339 N, 30 56 761 E	38	34	4	38	0
AY140	36 51 333 N, 30 56 757 E	58	52	6	43	15
AY161	36 51 339 N, 30 56 750 E	53	34	19	53	0
AY48	36 51 339 N, 30 56 717 E	60	57	3	58	2
AY93	36 51 340 N, 30 56 673 E	60	54	6	60	0
AY58	36 51 340 N, 30 56 636 E	44	28	16	44	0
AY26	36 51 339 N, 30 56 622 E	74	27	47	45	29
AY129	36 51 318 N, 30 56 479 E	32	26	6	24	8
AY81	36 51 296 N, 30 56 379 E	19	7	12	9	10
AY25	36 51 286 N, 30 56 356 E	42	6	36	6	36
AY92	36 51 292 N, 30 56 080 E	17	10	7	10	7
AY111	36 51 277 N, 30 56 061 E	33	30	3	33	0
AY110	36 51 277 N, 30 55 973 E	39	33	6	39	0
AY3	36 51 267 N, 30 55 812 E	40	32	8	40	0
AY49	36 51 265 N, 30 55 723 E	31	30	1	30	1
AY24	36 51 266 N, 30 55 719 E	60	56	4	37	23
AY104	36 51 261 N, 30 55 709 E	50	49	1	38	12
AY39	36 51 263 N, 30 55 667 E	22	18	4	22	0
AY124	36 51 257 N, 30 55 644 E	28	23	5	24	4
AY156	36 21 258 N, 30 55 627 E	20	19	1	18	2
AY130	36 51 243 N, 30 55 557 E	56	54	2	51	5
AY37	36 51 243 N, 30 55 450 E	47	44	3	47	0
AY1	36 51 243 N, 30 55 418 E	32	15	17	7	25
AY14	36 51 223 N, 30 55 353 E	9	0	9	3	6

nest ID	latitude/ longitude	total tracks	oriented	disoriented	reached sea	died
AOY44	36 51 145 N, 31 02 639 E	60	15	45	60	0
AOY34	36 51 155 N, 31 02 598 E	51	1	50	51	0
AOY10	36 51 158 N, 31 02 571 E	no data				
AOY27	36 51 162 N, 31 02 566 E	57	0	57	57	0
AOY2	36 51 184 N, 31 02 480 E	1	1	0	1	0
AOY32	36 51 175 N, 31 02 472 E	41	22	19	39	2
AOY18	36 51 194 N, 31 02 459 E	12	12	0	12	0
AOY42	36 51 188 N, 31 02 436 E	59	58	1	58	1
AOY22	36 51 202 N, 31 02 425 E	56	56	0	51	5
AOY3	36 51 205 N, 31 02 406 E	14	14	0	14	0
AOY33	36 51 205 N, 31 02 389 E	66	56	10	66	0
AOY36	36 51 213 N, 31 02 355 E	no data				
AOY13	36 51 219 N, 31 02 319 E	20	20	0	20	0
AOY29	36 51 237 N, 31 02 277 E	28	14	14	28	0
AOY31	36 51 239 N, 31 02 218 E	40	40	0	40	0
AOY19	36 51 242 N, 31 02 207 E	70	70	0	70	0
AOY28	36 51 244 N, 31 02 199 E	41	41	0	41	0
AOY20	36 51 244 N, 31 02 185 E	25	25	0	25	0
AOY5	36 51 250 N, 31 02 151 E	26	26	0	26	0
AOY35	36 51 254 N, 31 02 102 E	41	41	0	40	1
AOY41	36 51 260 N, 31 02 049 E	29	29	0	29	0
AOY38	36 51 263 N, 31 02 047 E	4	4	0	3	1
AOY45	36 51 264 N, 31 02 023 E	70	70	0	64	6
AOY6	36 51 265 N, 31 02 021 E	68	68	0	64	4
AOY7	36 51 266 N, 31 02 003 E	28	28	0	28	0
AOY39	36 51 275 N, 31 01 938 E	81	31	50	78	3
AOY37	36 51 279 N, 31 01 894 E	72	54	18	72	0
AOY8	36 51 287 N, 31 01 798 E	70	38	32	70	0
AOY24	36 51 289 N, 31 01 783 E	44	29	15	44	0
AOY46	36 51 292 N, 31 01 761 E	70	40	30	67	3
AOY9	36 51 304 N, 31 01 698 E	28	28	0	28	0
AOY30	36 51 327 N, 31 01 370 E	75	44	31	73	2
AOY25	36 51 326 N, 31 01 305 E	3	2	1	2	1
AOY14	36 51 447 N, 31 00 539 E	55	55	0	55	0
AOY17	36 51 451 N, 31 00 506 E	no data				
AOY15	36 51 479 N, 30 59 904 E	74	24	50	74	0
AOY26	36 51 487 N, 30 59 814 E	40	40	0	40	0
AOY1	36 51 482 N, 30 59 758 E	41	0	41	41	0
AOY12	36 51 486 N, 30 59 747 E	70	0	70	70	0
AOY11	36 51 474 N, 30 59 679 E	no data				
AOY21	36 51 475 N, 30 59 656 E	39	20	19	39	0
AOY23	36 51 476 N, 30 59 636 E	60	60	0	60	0
AOY43	36 51 472 N, 30 59 482 E	60	0	60	60	0
AOY16	36 51 472 N, 30 58 613 E	70	0	70	70	0

nest ID	latitude, longitude	total tracks	oriented	disoriented	reached sea	died
OY43	36 50 395 N, 31 05 875 E	43	0	43	0	43
OY45	36 50 440 N, 31 05 669 E	99	38	61	88	11
OY24	36 50 453 N, 31 05 601 E	no data				
OY40	36 50 452 N, 31 05 598 E	no data				
OY47	no data	no data				
OY17	36 50 521 N, 31 04 659 E	no data				
OY30	36 50 532 N, 31 04 632 E	no data				
OY32	36 50 566 N, 31 04 490 E	no data				
OY21	36 50 581 N, 31 04 481 E	26	26	0	26	0
OY27	36 50 534 N, 31 04 476 E	no data				
OY41	36 50 685 N, 31 04 359 E	28	25	3	28	0
OY42	no data	no data				
OY35	36 50 707 N, 31 04 316 E	73	73	0	73	0
OY16	36 50 739 N, 31 04 266 E	40	18	22	31	9
OY59	36 50 746 N, 31 04 243 E	87	40	47	87	0
OY7	36 50 750 N, 31 04 236 E	23	21	2	20	3
OY57	36 50 761 N, 31 04 219 E	67	60	7	67	0
OY15	36 50 767 N, 31 04 207 E	13	11	2	13	0
OY51	36 50 773 N, 31 04 192 E	74	56	18	68	6
OY20	36 50 782 N, 31 04 192 E	47	37	10	41	6
OY58	36 50 770 N, 31 04 192 E	70	40	30	65	5
OY5	36 50 732 N, 31 04 188 E	23	13	10	23	0
OY4	36 50 791 N, 31 04 175 E	50	47	3	50	0
OY3	36 50 792 N, 31 04 172 E	30	28	2	22	8
OY33	36 50 804 N, 31 04 159 E	no data				
OY18	36 50 809 N, 31 04 151 E	no data				
OY8	36 50 816 N, 31 04 121 E	no data				
OY44	36 50 817 N, 31 04 105 E	63	62	1	59	4
OY11	36 50 323 N, 31 04 101 E	74	74	0	74	0
OY9	36 50 227 N, 31 04 071 E	13	9	4	12	1
OY48	36 50 839 N, 31 04 057 E	no data				
OY12	36 30 840 N, 31 04 035 E	44	44	0	43	1
OY38	36 50 857 N, 31 04 013 E	no data				
OY13	36 50 844 N, 31 04 003 E	37	37	0	37	0
OY2	36 50 859 N, 31 03 991 E	no data				
OY10	36 50 381 N, 31 03 956 E	64	64	0	64	0
OY22	36 50 871 N, 31 03 939 E	38	38	0	38	0
OY46	36 50 887 N, 31 03 923 E	no data				
OY1	36 50 889 N, 31 03 909 E	75	13	62	75	0
OY56	36 50 900 N, 31 03 872 E	no data				
OY28	36 50 898 N, 31 03 851 E	51	51	0	51	0
OY54	36 50 908 N, 31 03 825 E	20	20	0	20	0
OY36	36 50 948 N, 31 03 701 E	61	61	0	61	0
OY39	36 50 959 N, 31 03 679 E	68	60	8	68	0
OY34	36 50 951 N, 31 03 671 E	54	53	1	53	1
OY25	36 50 972 N, 31 03 599 E	65	65	0	65	0
OY31	36 51 054 N, 31 03 262 E	41	41	0	41	0
OY6	36 51 052 N, 31 03 250 E	no data				
OY37	36 51 078 N, 31 03 159 E	77	65	12	73	4
OY29	36 51 070 N, 31 03 147 E	41	18	23	41	0
OY14	36 51 087 N, 31 03 095 E	no data				
OY52	36 51 091 N, 31 03 009 E	39	39	0	39	0
OY49	36 51 101 N, 31 02 957 E	50	17	33	50	0
OY26	36 51 112 N, 31 02 901 E	44	44	0	44	0
OY50	36 51 129 N, 31 02 812 E	61	52	9	55	6
OY23	36 51 130 N, 31 02 807 E	no data				
OY19	36 51 129 N, 31 02 807 E	90	90	0	90	0

Section I, Western and Central areas

Central Area
ND: 33.5
HD: 86.5
HM: 47.9

Western area
ND: 29.5
HD: 54.7
HM: 78.9

Fishing vessels

Small settlement

Besgöz River mouth

Small huts

ALSD: 0

Note high number of disoriented hatchlings due to long distance light propagation (> 2 km)

Hotel area

Aksu River mouth

1 km
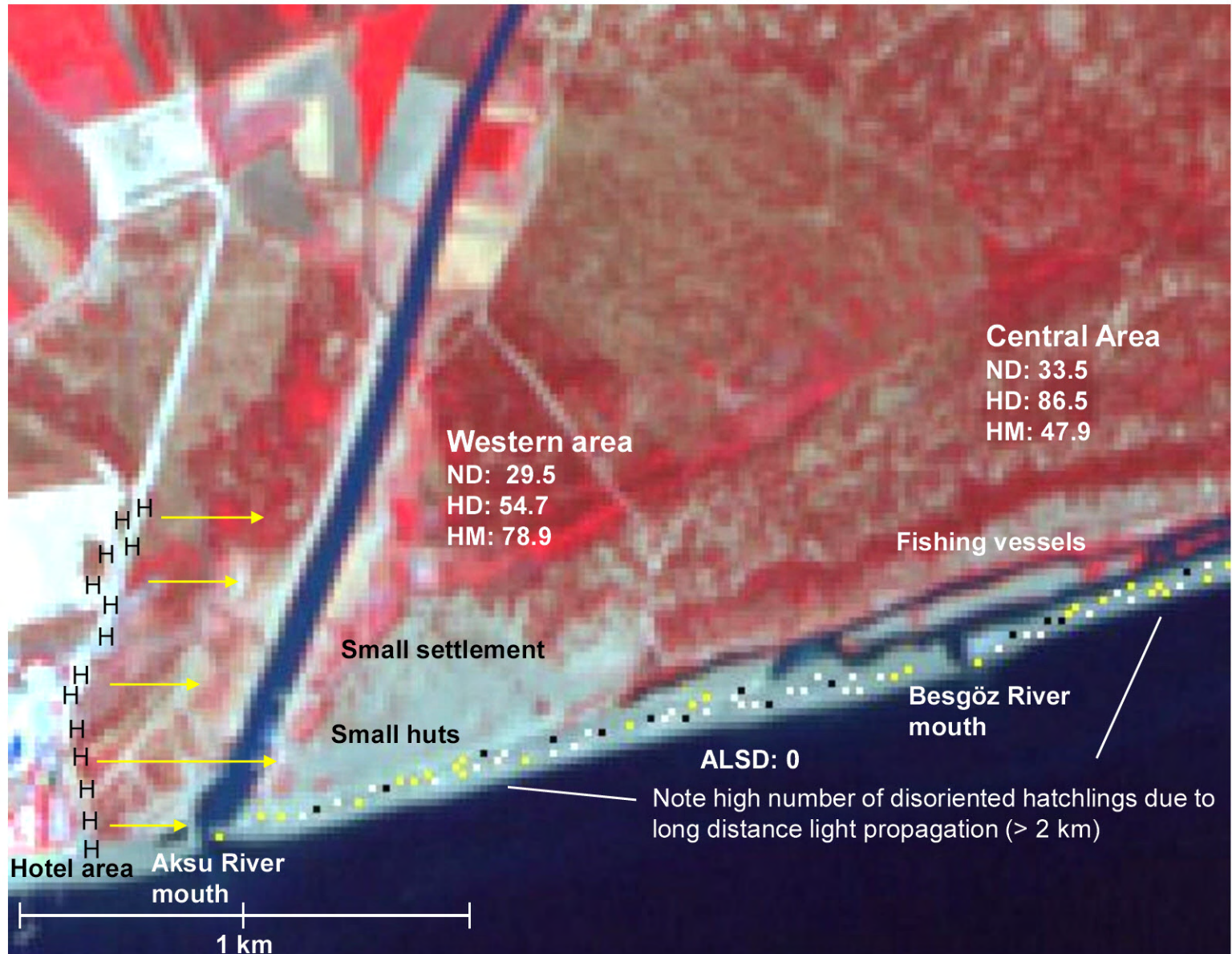
APPENDIX 3 a) Distribution of *C. caretta* nests in Section I. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation). ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

Section I, Central and Eastern areas

Eastern Area
ND: 24.5
HD: 6.1
HM: 0

Central Area
ND: 33.5
HD: 86.5
HM: 47.9

Besgöz River

H

Hotel Asteria

Note lower nest density
close to Hotel zone 1

ALSD: 0

High nest density and low hatchling disorientation
Point of lowest illuminance measured (< 0.11 lux)

ALSD: 0

Disoriented hatchlings still
found in area of low degree
coastal development due to light
propagation

1 km
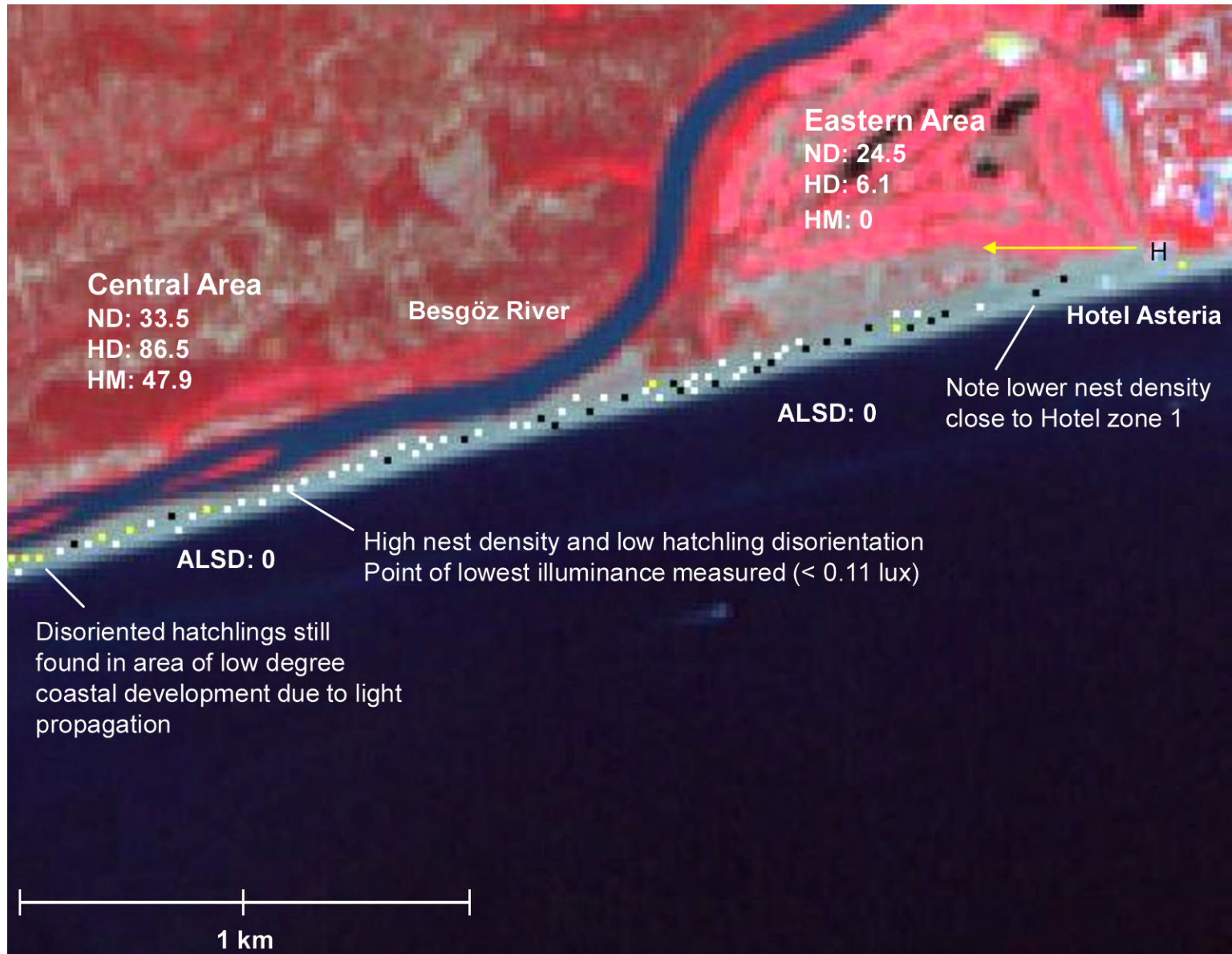
APPENDIX 3 b) Distribution of *C. caretta* nests in Section I. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation). ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

APPENDIX 3 c) Distribution of *C. caretta* nests in Section II. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation).
ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

Section II, Hotel zone 2 and Public area 2

Hotel zone 2
ND: 3
HD: 62.6
HM: 0

Public area 2
ND: 20
HD: 197
HM: 3.4

Construction site

Hotel Justiniano
Sun Zeynep
Club Justiniano
Altis
Papillon Belvil
Marmara Bellis
Attelia
Sand dune

ALSD: 6.3

ALSD: 1.5

Note high ALSD reducing nesting density in Hotel zone 2

Note high nest number and reduced hatchling disorientation in front of an artificial sand dune (light cover)

Note high nesting density but also increased hatchling disorientation in Public area 2 (light trapping)

1km

APPENDIX 3 d) Distribution of *C. caretta* nests in Section II. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation).
ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

Section II, Hotel zone 3 and Public area 3

Hotel zone 3
ND: 8.5
HD: 48.9
HM: 17.2

Public area 3
ND: 19.5
HD: 121.2
HM: 6.3

Magic Life

Belconti

Arcadia

Grida

Ali Bey

H   H   H   H   H   H   H   H

ALSD: 5.1

Sand dunes
Construction site

Note Metal Halide lamps located in
Hotel zone 3 affecting Public area 3
(light trapping)

1km
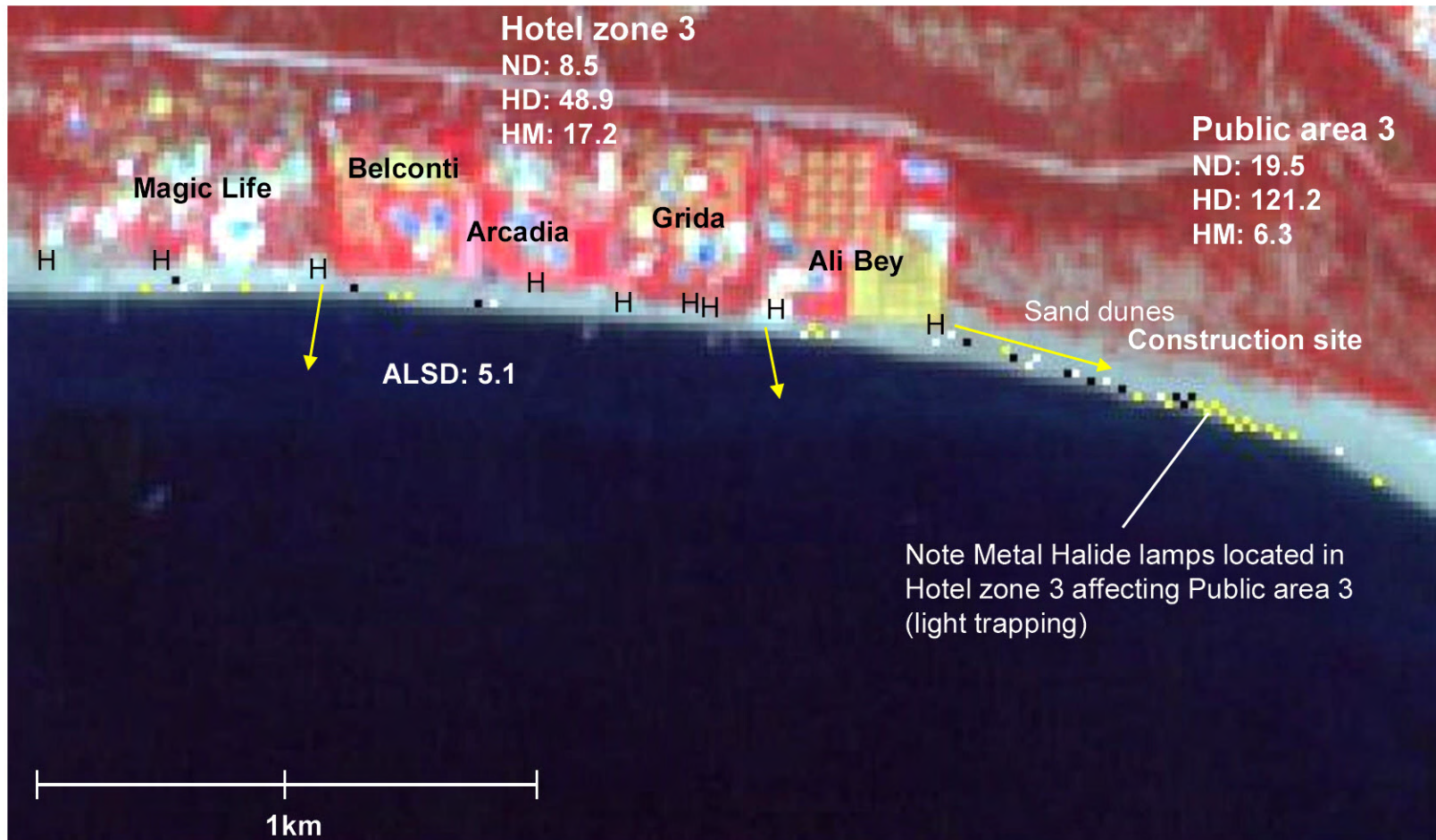
APPENDIX 3 e) Distribution of *C. caretta* nests in Section II. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation). ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

Section II, Public area 3 and Hotel zone 4

Public area 3
ND: 19.5
HD: 121.2
HM: 6.3

Coastal street

Note low nesting density in front of HPS lights

HPS lights

Sillyum

Cornelia

Pine Beach

Gloria Verde

Atlantis   Zeugma

Hotel Zone 4
ND: 2.5
HD: 66.7
HM: 23.4

Cesar's Temple

Rixos Premium

ALSD: 5.1

ALSD: 9

Note high ALSD. Metal Halide lamps (1000 watts) positioned 10 m from sea turtle nest.
Point of highest illuminance measured (40 lux)

Acisu River mouth

1 km

APPENDIX 3 f) Distribution of *C. caretta* nests in Section II. White dots: nests without disoriented hatchlings, yellow dots: nests showing disoriented hatchlings, black dots: Data deficient. H: Metal Halide lights (yellow arrows indicate light propagation). Numbers indicate the position of 16 High Pressure Sodium Vapour (HPS) lights along a coastal street.
ND: Nest density/km, HD: hatchling disorientation/km, HM: hatchling mortality/km, ALSD: Artificial Light Source Density

Appendix 4

Hatchling orientation in the circular arena in Xcacel (Chapter 5, Setup 2)

Digits indicate the number of hatchlings and their final position in the arena. Digits in the centre of the circle indicate inactive hatchlings, which did not move. Treatment groups according to Tables 5.2, 5.3 and 5.4.

MV: mean vector, LMV: length of mean vector.

Deviations from normal (MV ± 20°, LMV < 0.9) in **bold**

Control C. mydas — 7 | 8; MV: 1.6º; LMV: 0.95

400 Hz (40.8 dB) C. mydas — 7 | 8; MV: 1.6º; LMV: 0.95
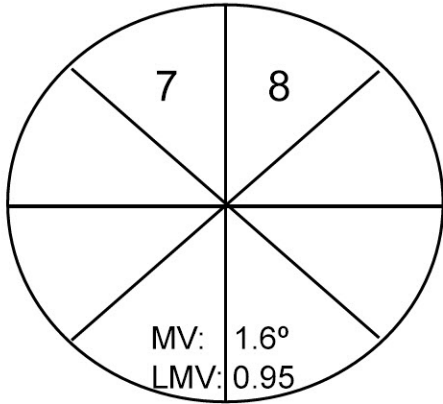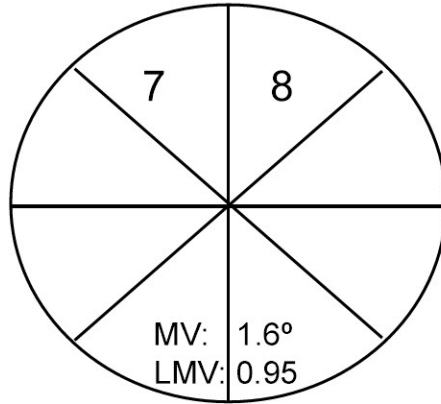
1000 Hz (45.6 dB) C. mydas — 8 | 8; MV: 0º; LMV: 0.95
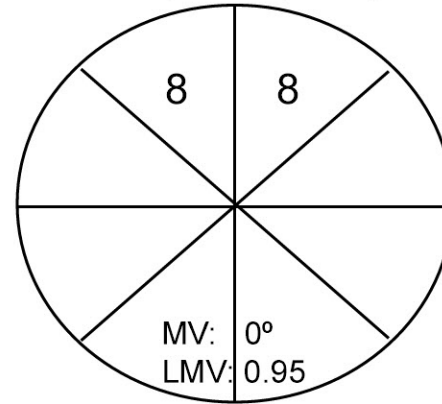
Wave sound (44 dB) C. mydas — 8 | 8; MV: 0 º; LMV: 0.95
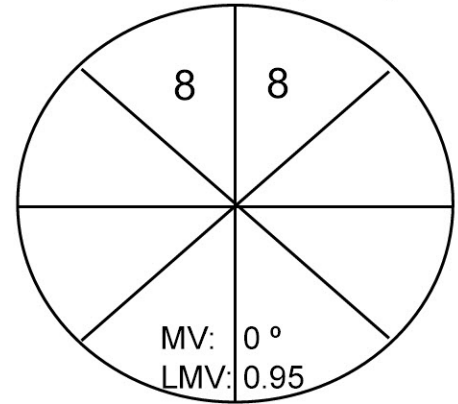
Surf sound (44.6 dB) C. mydas — 4 | 12; MV: 11.7º; LMV: 0.97
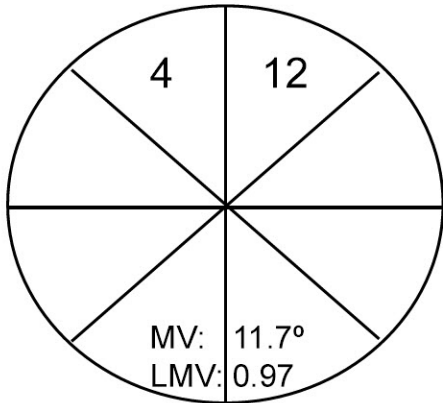
Pink Noise (41.6 dB) C. mydas — 7 | 11; MV: 5.3º; LMV: 0.95
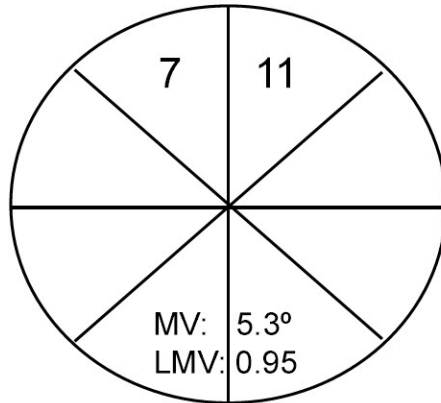
Fireworks (55.4 dB) C. mydas — 2 | 12; MV: 16.5º; LMV: 0.95
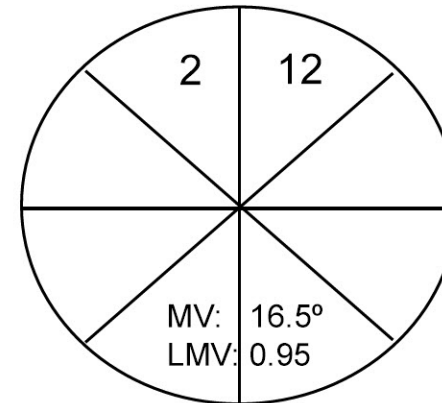
Street traffic (47.2 dB) C. mydas — 8 | 8; MV: 0º; LMV: 0.95

400 Hz (50.6 dB) C. mydas — 10 | 6; MV: 354º; LMV: 0.95
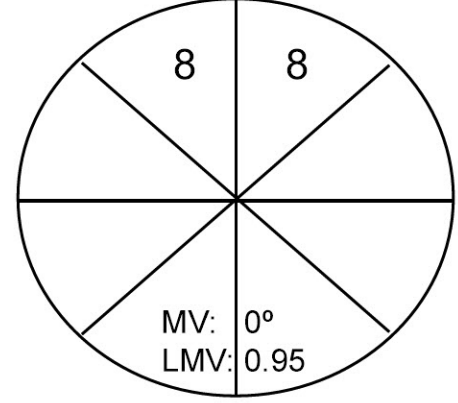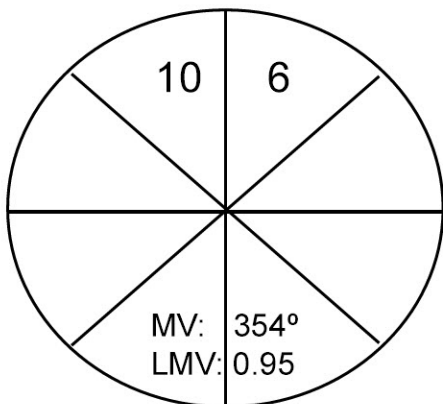
1000 Hz (55.7 dB) C. mydas — 7 | 7 | 1; MV: 0º; LMV: 0.95
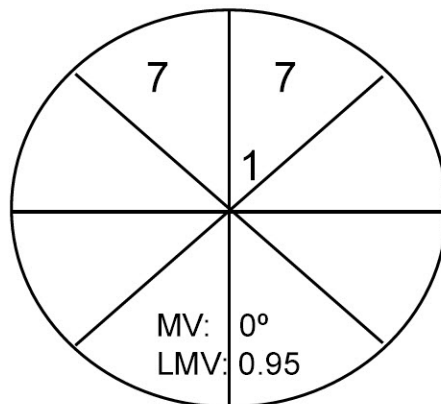
Wave sound (55.4 dB) C. mydas — 9 | 4 | 2 | 1; **MV: 339.3º**; **LMV: 0.85**

Surf sound (55.1 dB) C. mydas — 8 | 9; MV: 1.4º; LMV: 0.95
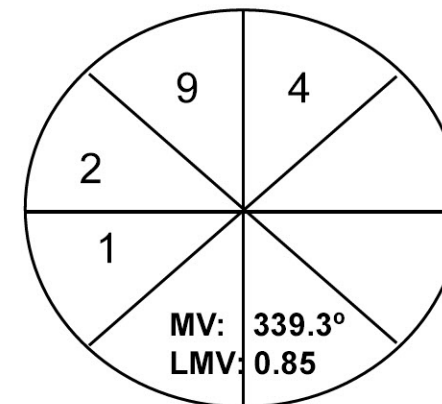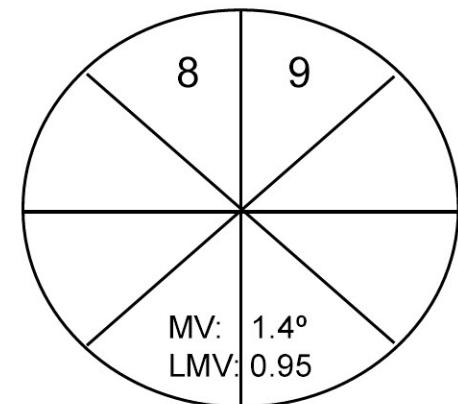
Control C. caretta
11 | 5
MV: 351,2°
LMV: 0.96
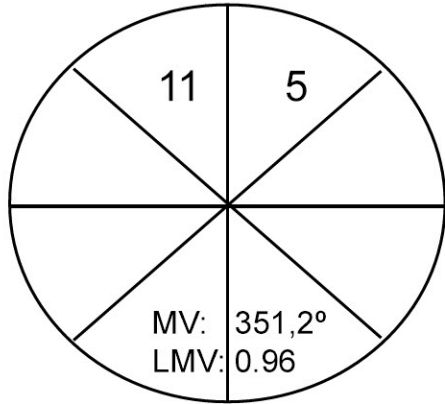
400 Hz (40.8 dB) C. caretta
6 | 8
MV: 3.4°
LMV: 0.95
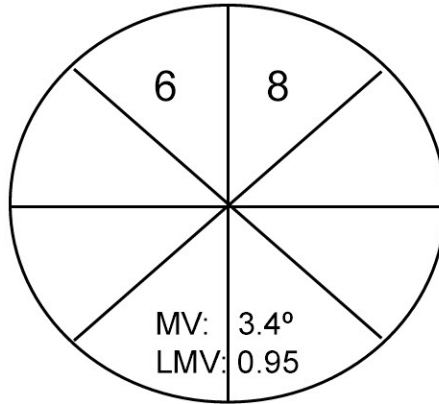
1000 Hz (45.6 dB) C. caretta
8 | 4
MV: 352°
LMV: 0.96
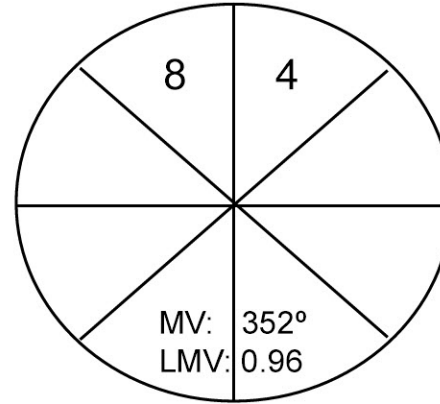
Wave sound (44 dB) C. caretta
7 | 8
1
MV: 1.6°
LMV: 0.95
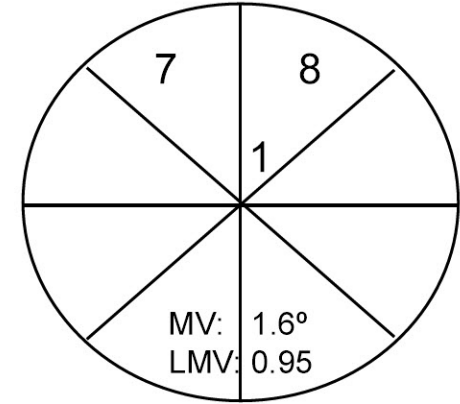
Surf sound (44.6 dB) C. caretta
6 | 8
2
MV: 3.4°
LMV: 0.95
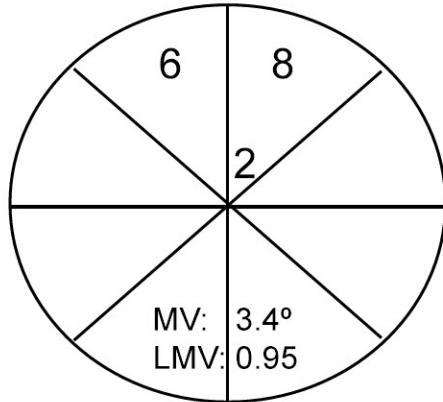
Pink noise (41.6 dB) C. caretta
10 | 6
MV: 354.1°
LMV: 0.95
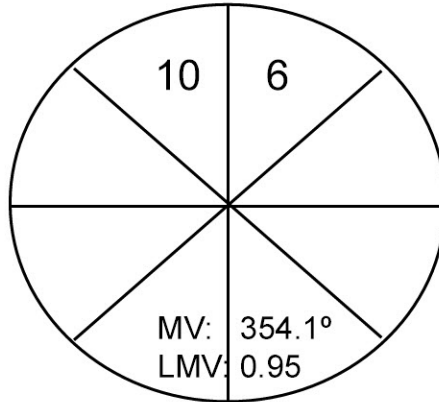
Fireworks (55.4 dB) C. caretta
9 | 4
3
MV: 351°
LMV: 0.96
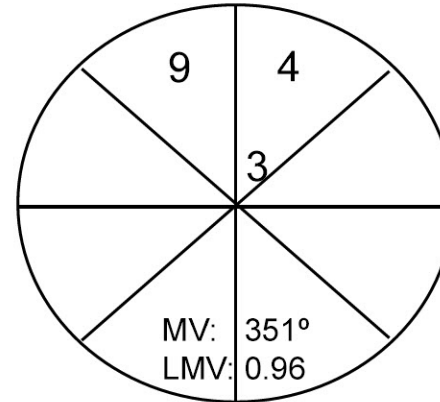
Street traffic (47.2 dB) C. caretta
6 | 9
2
MV: 4.7°
LMV: 0.95
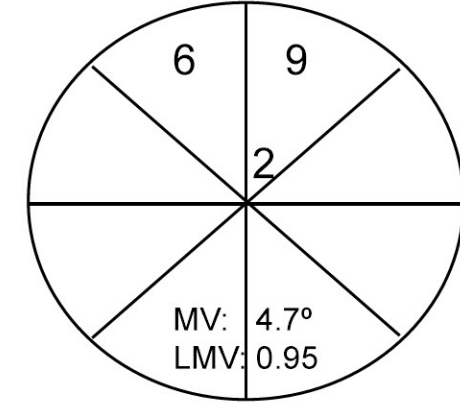
400 Hz (50.6 dB) C. caretta
7 | 4
3 | 2
MV: 340.9°
LMV: 0.88
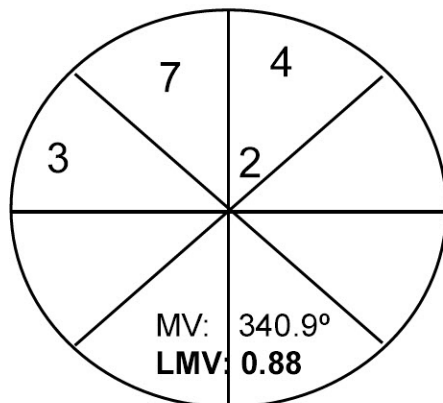
1000 Hz (55.7 dB) C. caretta
7 | 8
1
MV: 357.8°
LMV: 0.91
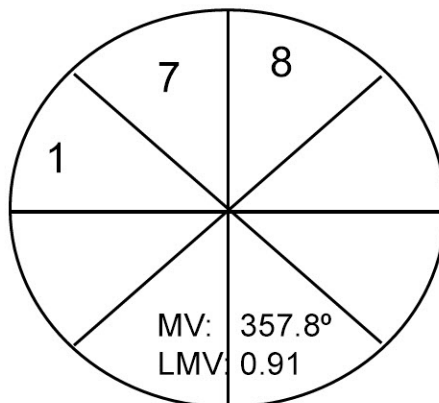
Wave sound (55.4 dB) C. caretta
4 | 10
2
MV: 10.1°
LMV: 0.96
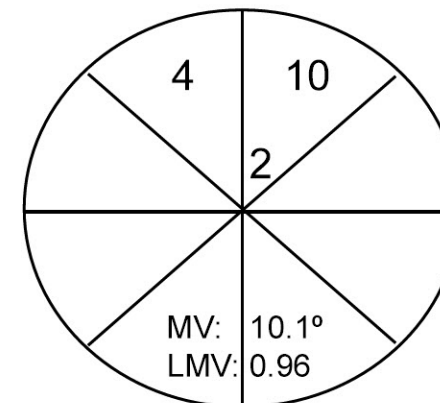
Surf sound (55.1 dB) C. caretta
7 | 7
2
MV: 0°
LMV: 0.95

White (7.9 lux) C. mydas — 7, 8; MV: 1.6°, LMV: 0.95

White (27.6 lux) C. mydas — 7, 9; MV: 3°, LMV: 0.95

UV (2.1 lux) C. mydas — 4, 4, 6, 2; **MV: 320.4°, LMV: 0.74**

UV (3.75 lux) C. mydas — 5, 4, 4, 3; **MV: 321.8°, LMV: 0.71**
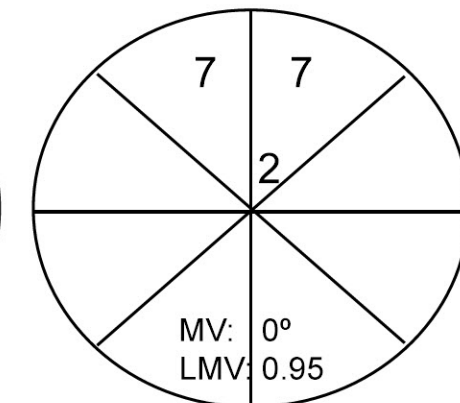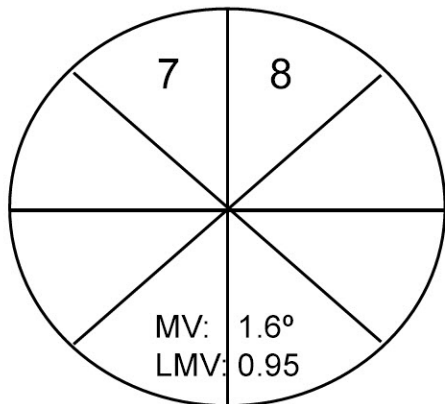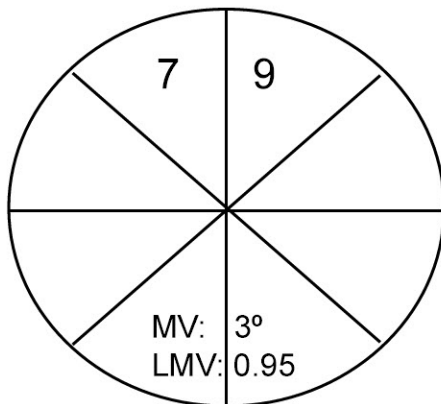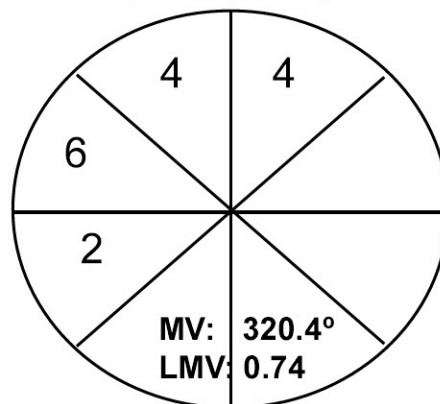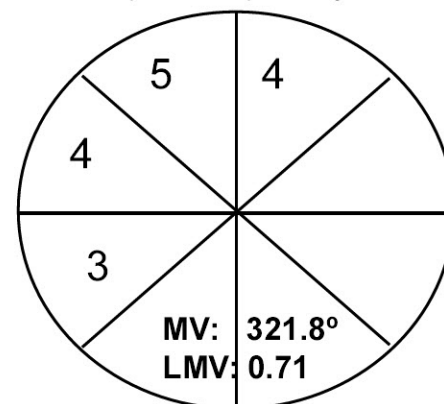
Magenta (4.4 lux) C. mydas — 15, 1; **MV: 25.1°**, LMV: 1

Magenta (9.2 lux) C. mydas — 7, 9; MV: 3°, LMV: 0.95
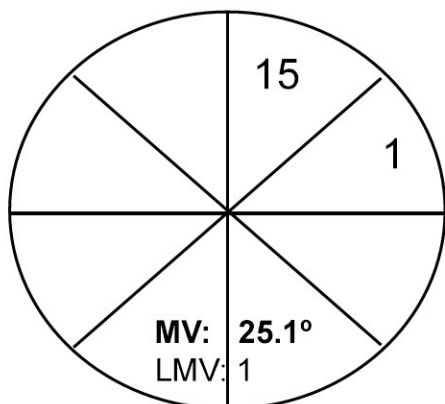
Green (3.3 lux) C. mydas — 3, 10, 2, 8; **MV: 33.7°, LMV: 0.89**

Green (10.2 lux) C. mydas — 5, 11; MV: 8.8°, LMV: 0.96
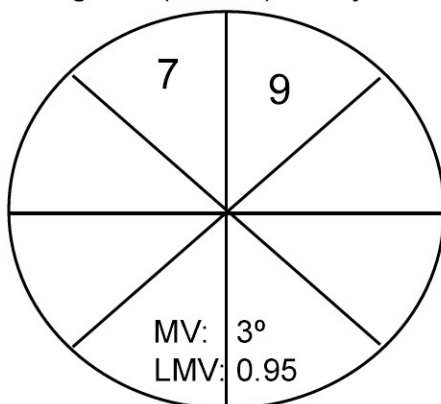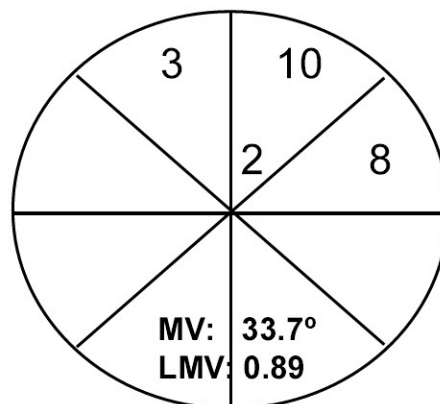
Yellow (7 lux) C. mydas — 6, 10; MV: 5.9°, LMV: 0.95

Yellow (21.5 lux) C. mydas — 11, 4, 1; MV: 345.8°, LMV: 0.94

Red (4.5 lux) C. mydas — 8, 8; MV: 0°, LMV: 0.95

Red (9.6 lux) C. mydas — 6, 9, 1; MV: 3.4°, **LMV: 0.83**
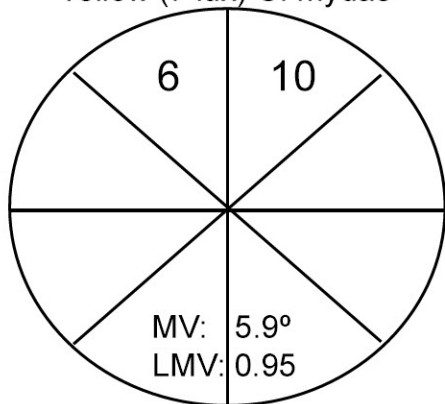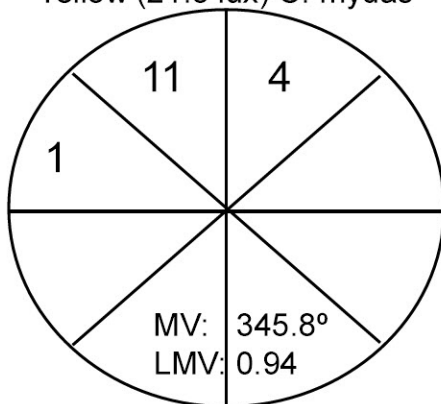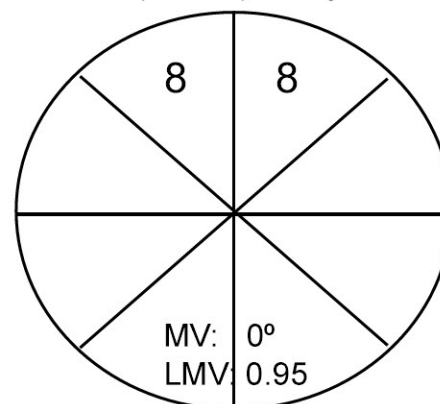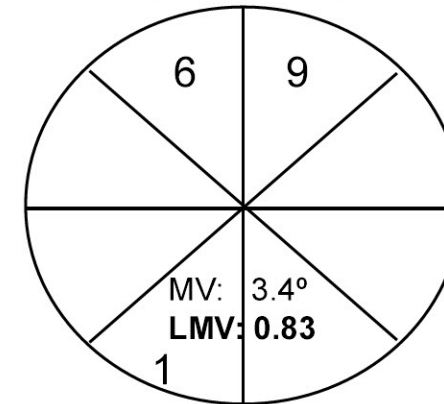
White (7.9 lux) C. caretta
4 | 9
MV: 9.1°
LMV: 0.96

White (27.6 lux) C. caretta
3 | 5
8
MV: 5.9°
LMV: 0.95
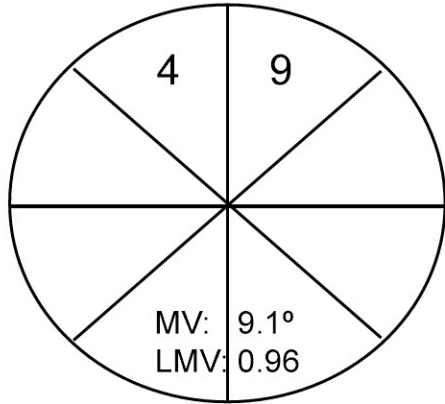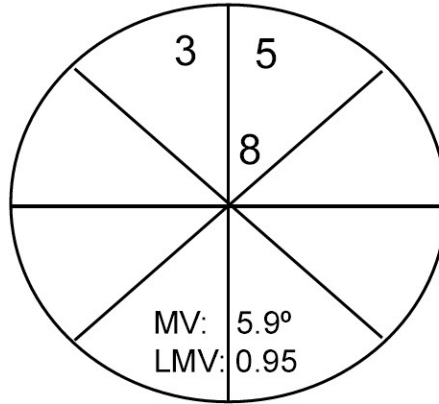
UV (2.1 lux) C. caretta
8 | 6
1
1
MV: 348.6°
LMV: 0.85
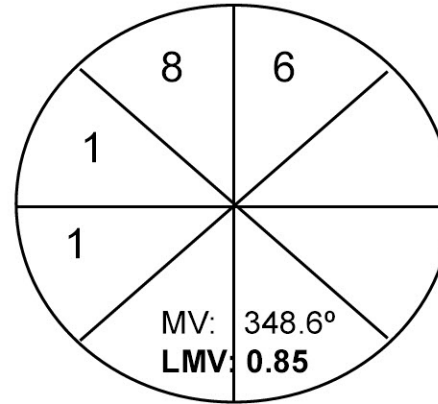
UV (3.75 lux) C. caretta
6
6
2
1 **MV: 301.4°**
LMV: 0.81
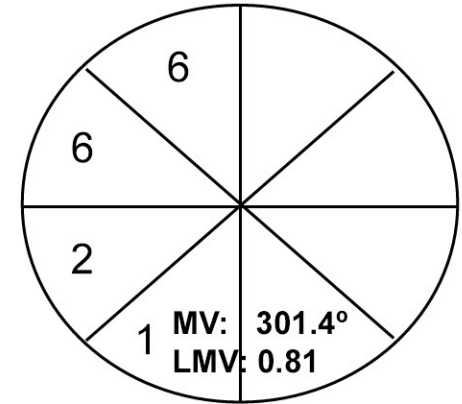
Magenta (4.4 lux) C. caretta
9 | 6
MV: 355.3°
LMV: 0.95
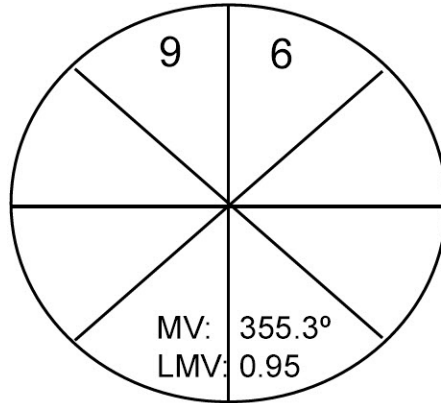
Magenta (9.2 lux) C. caretta
7 | 7
MV: 0°
LMV: 0.95
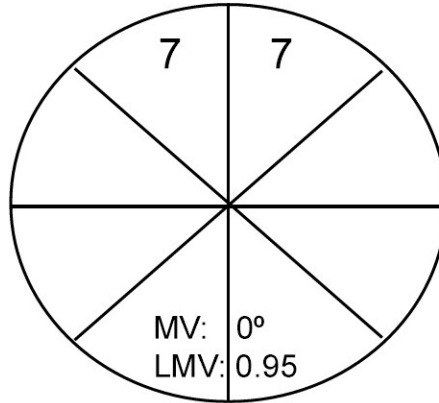
Green (3.3 lux) C. caretta
5 | 11
7 | 10
MV: 31.8°
LMV: 0.86
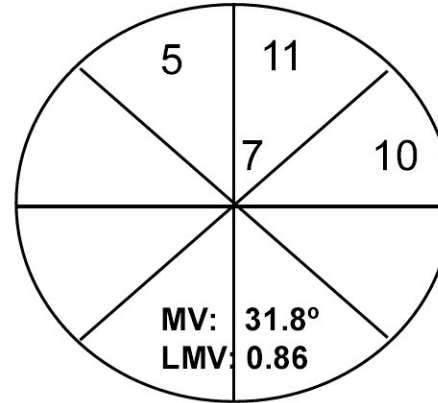
Green (10.2 lux) C. caretta
4 | 5
MV: 2.6°
LMV: 0.95
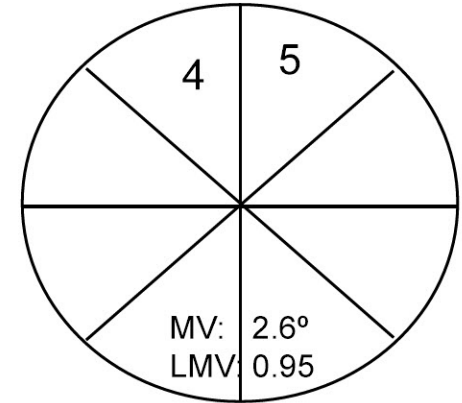
Yellow (7 lux) C. caretta
8 | 5
2
1
MV: 342.5°
LMV: 0.83
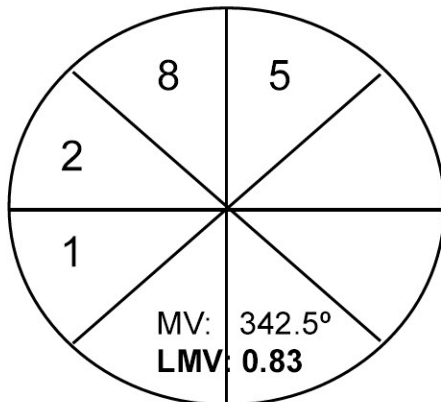
Yellow (21.5 lux) C. caretta
10 | 3
3
MV: 337.5°
LMV: 0.91

Red (4.5 lux) C. caretta
9 | 6
MV: 355.3°
LMV: 0.95
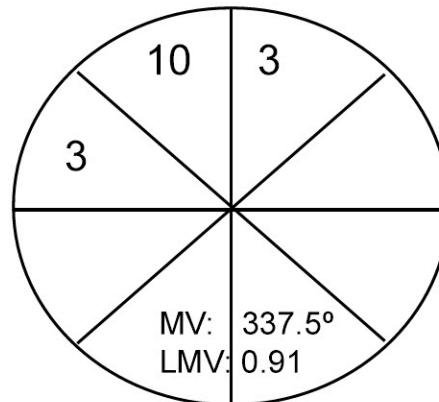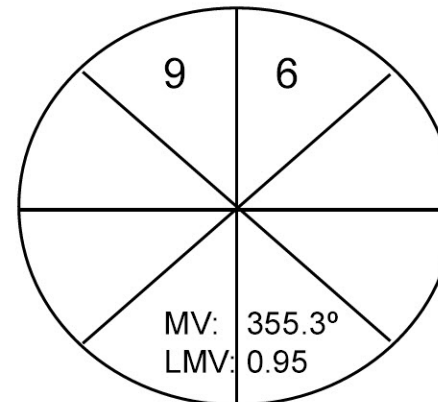
Red (9.6 lux) C. caretta
8 | 8
MV: 0°
LMV: 0.95
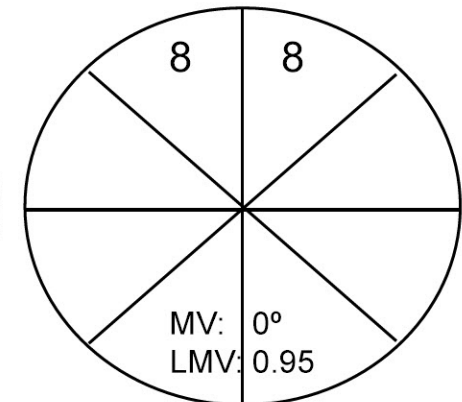
White vs. Wave sound C. mydas

2 | 14

MV: 17.3º
LMV: 0.99

White vs. Surf sound C. mydas

8 | 8

MV: 0º
LMV: 0.95
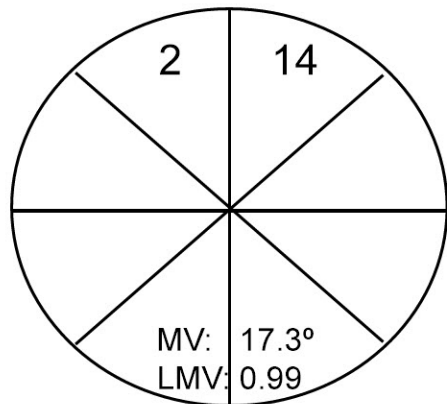
UV vs. Wave sound C. mydas

8 | 8

MV: 0º
LMV: 0.95
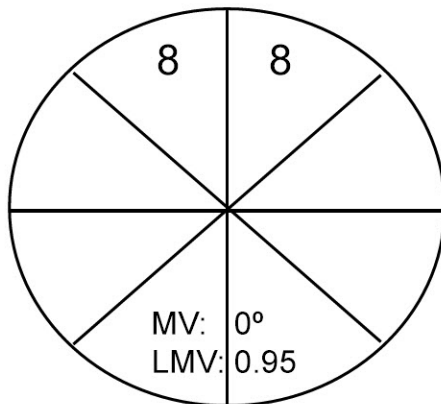
UV vs. Surf sound C. mydas

6
4
4
1 | 1

MV: 343.7º
LMV: 0.52
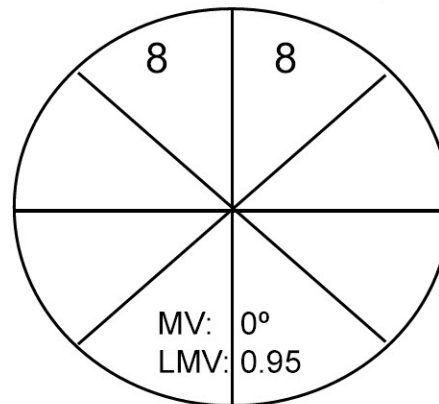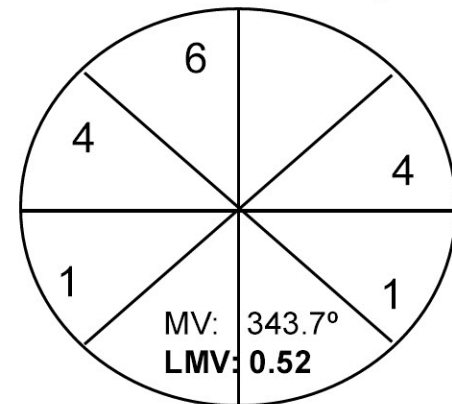
Magenta vs. Wave sound C. mydas

6 | 9
1

MV: 1º
LMV: 0.86

Magenta vs. Surf sound C. mydas

5 | 8
3

MV: 351.5º
LMV: 0.7

Green vs. Wave sound C. caretta

7 | 6
1

MV: 358.2º
LMV: 0.95
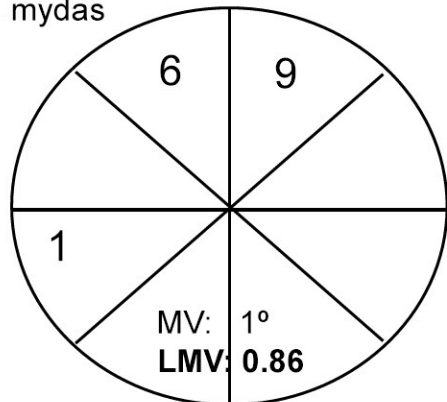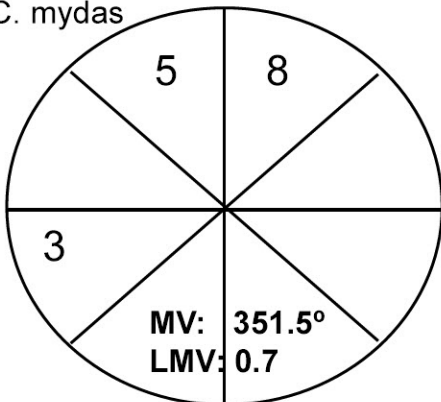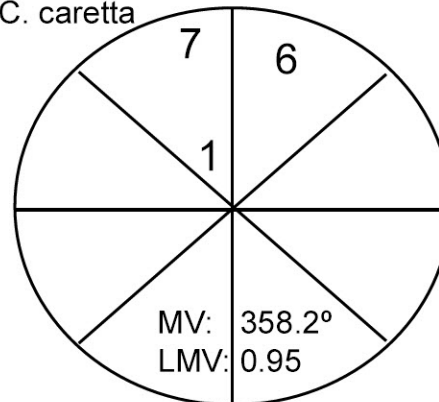
Green vs. Surf sound C. mydas

16

MV: 22.5º
LMV: 1
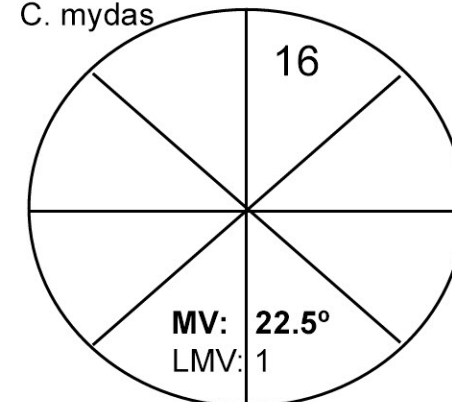
Yellow vs. Wave sound C. mydas

7 | 5
4

MV: 356.1º
LMV: 0.95
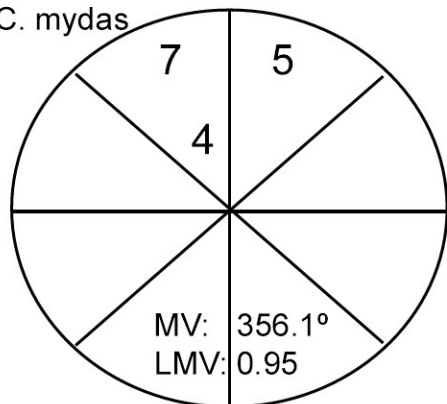
Yellow vs. Surf sound C. mydas

8 | 8

MV: 0º
LMV: 0.95
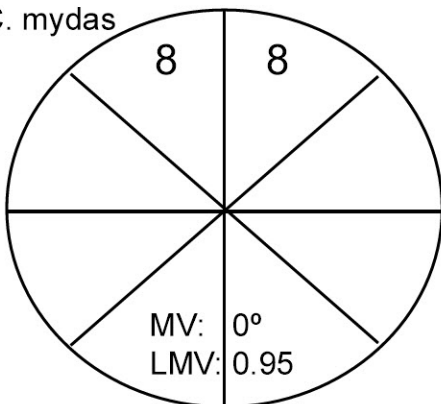
Red vs. Wave sound C. mydas

Not done
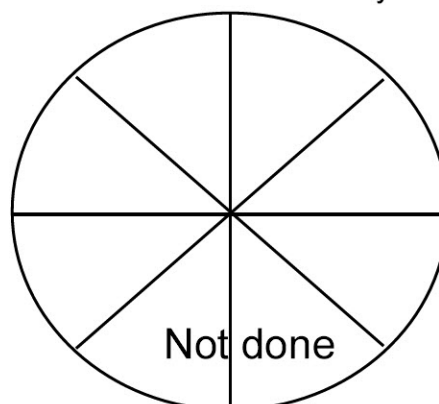
Red vs. Surf sound C. mydas

8 | 9

MV: 1.4º
LMV: 0.95
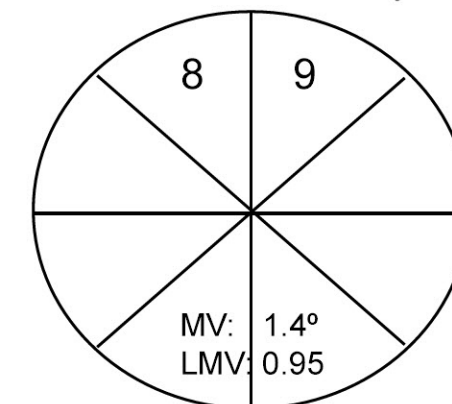
White vs. Wave sound C. caretta
8 | 8
MV: 0º
LMV: 0.95

White vs. Surf sound C. caretta
10 | 6
MV: 354.1º
LMV: 0.95

White vs. 1000 Hz C. caretta
7 | 9
MV: 3º
LMV: 0.95
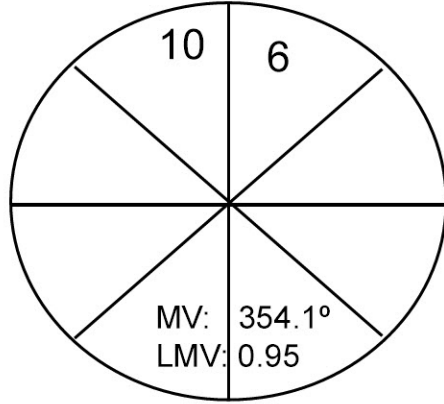
UV vs. Surf sound C. caretta
7 | 1
7
MV: 319º
LMV: 0.913

Magenta vs. Wave sound C. caretta
5 | 11
MV: 8.8º
LMV: 0.96

Magenta vs. Surf sound C. caretta
6 | 10
MV: 6º
LMV: 0.95
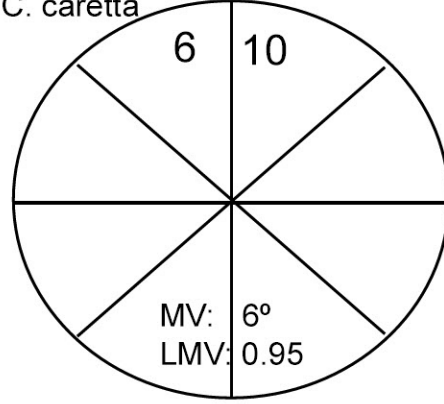
Green vs. Wave sound C. caretta
7 | 6
1
MV: 358.2º
LMV: 0.95
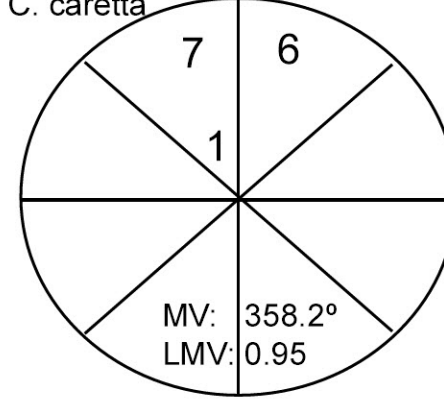
Green vs. Surf sound C. caretta
8 | 8
MV: 0º
LMV: 0.95
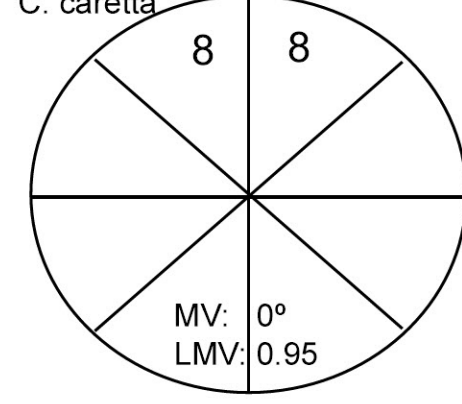
Yellow vs. Wave sound C. caretta
Not done
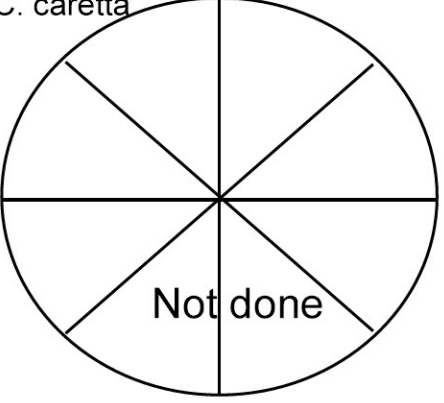
Yellow vs. Surf sound C. caretta
Not done
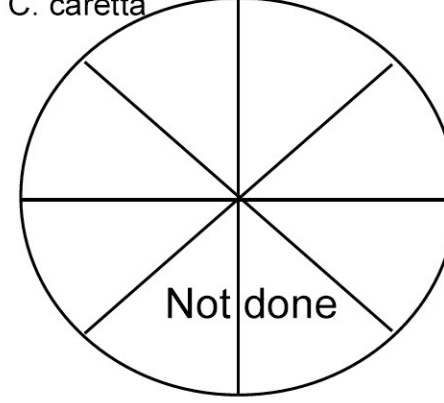
Red vs. Wave sound C. caretta
7 | 9
MV: 3º
LMV: 0.95
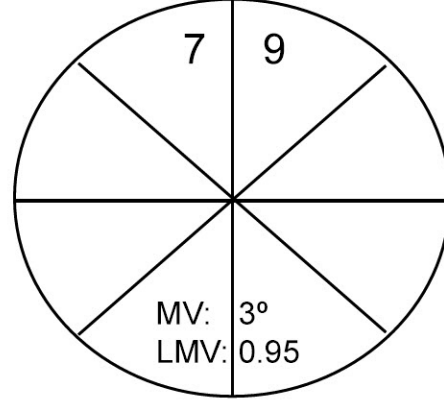
Red vs. Surf sound C. caretta
8 | 8
MV: 0º
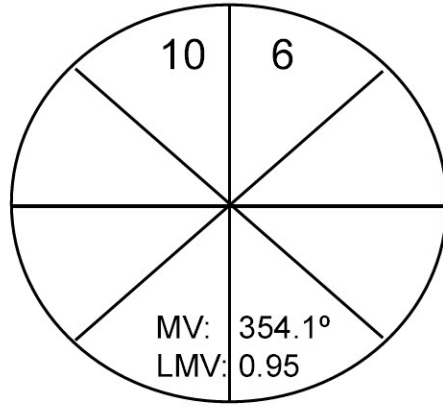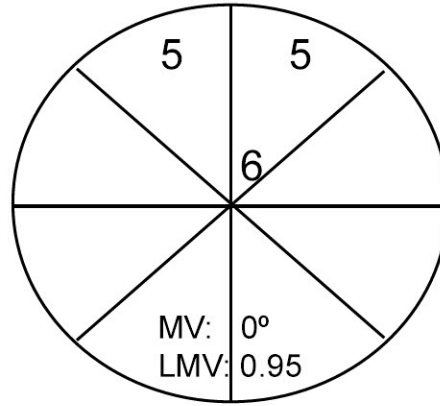LMV: 0.95
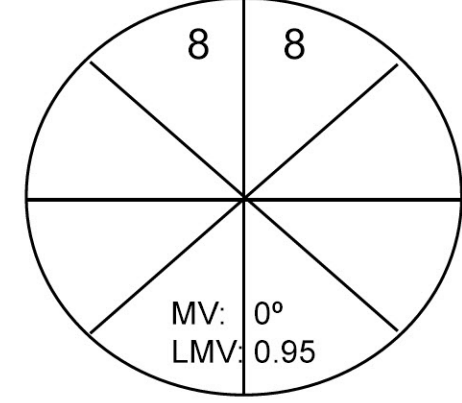
| White vs. 400 Hz C. mydas | White vs. 1000 Hz C. mydas | White vs. 400 Hz C. caretta | White vs. 1000 Hz C. caretta |

White vs. 400 Hz C. mydas
7 9
MV: 3°
LMV: 0.95

White vs. 1000 Hz C. mydas
10 6
MV: 354.1°
LMV: 0.95

White vs. 400 Hz C. caretta
5 5
6
MV: 0°
LMV: 0.95

White vs. 1000 Hz C. caretta
7 9
MV: 3°
LMV: 0.95